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The Regional Implications of Transported Air Pollutants:

An Assessment of Acidic Deposition and Ozone

Interim Draft

This interim draft is a product of the ongoing OTA assessment of the "Impacts of Atmospheric Alteration." Available information and analysis is being released at this time to assist congressional deliberations on the Clean Air Act. This draft has neither been reviewed or approved by the Technology Assessment Board. Portions of this material will be contained in the forthcoming OTA Report on transport of air pollutants.

July 1982

CONGRESS OF THE UNITED STATES
Office of Technology Assessment
Washington, D.C. 20510

J952-30

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An Assessment of Acidic Deposition and Ozone

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FOREWORD

Acid rain has been the focus of much debate during the Clean Air Act deliberations of the 97th Congress. The controversy over transported air pollutants -- of which acid rain is one example -- revolves around suspected damage to lakes and streams, forests and soils, agricultural crops, building materials and human health. Some individuals and groups believe that more stringent measures should be taken now by the Congress to control pollutant emissions. Others, pointing to the uncertainties in our knowledge of the causes and consequences of transported air pollutants, are concerned that the costs of such actions may be unwarranted.

Recognizing that the uncertainty surrounding many aspects of the problem complicates the decision of whether, or when, to control transported air pollutants, the Committees responsible for reauthorizing the Clean Air Act asked OTA to evalute the potential implications of transported air pollutants, such as acid rain and ozone. In response to these immediate concerns of Congress, this OTA interim draft focuses on the potential benefits of acting now to abate transported air pollutants versus the potential costs of action that may be premature.

Current understanding of transported pollutants allows a description of potential resource damage -- regions of the U.S. most susceptible to damage -- but not a precise quantification. The costs of likely control strategies, and who might bear these costs, can be described, but the effectiveness of these control strategies for avoiding resource damage cannot be calculated with confidence. Given the difficulty of quantifying the relationship between emissions reductions and the prevention of resource damage, policy decisions may have to be based on the risks of resource damage, the risks of unwarranted control expenditures, and the distribution of these risks among different groups and regions of the country.

Throughout this interim draft, special attention is paid to the regional distribution of risks -- both to those who may bear potentially unnecessary control costs, as well as those who face potential resource damage. This draft is being made available to the 97th Congress to assist its current deliberations on the Clean Air Act. Policy options are not included in this interim draft, but will be presented in the final Report. The Report, to be completed later this year, will also include an updated version of information contained in this draft, additional research results, and a synthesis of findings.

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INTERIM DRAFT

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BB Allocation of Sulfur Dioxide Emissions Reductions and the Costs of Control

CC Control Technologies for Reducing Sulfur and Nitrogen Oxide Emissions

DD The Effects of Acid Rain Control on the U.S. Coal Market

FF 1) Effects of Ozone on Agricultural Productivity
2) Effects of Acid Rain on Agricultural Crops
3) Fate and Effects of Hydrogen Ions, Sulfate, and Nitrate in Forest Ecosystems
4) Effects of Ozone on Forest Productivity

GG Regional Assessment of Aquatic Resources at Risk from Acidic Deposition

HH 1) Health Effects of Sulfur Dioxide
2) Health Effects of Nitrogen Oxides
3) Indirect Health Effects of Acid Precipitation

MM 1) Interstate Provision of the Present Clean Air Act
2) Mechanisms for Control of Transboundary Pollution

I. INTRODUCTION

A. THE OTA ASSESSMENT

The Congressional Office of Technology Assessment (OTA) is analyzing the Impacts of Atmospheric Alterations at the request of the Senate Committee on Environment and Public Works and the House Committee on Energy and Commerce. The assessment is designed to synthesize the best available information on potential impacts of long-range transport air pollutants such as acid deposition and ozone.

The assessment focuses on the potential benefits of acting now to abate long-range transport air pollution versus the potential costs of action that may be premature. Specifically, the goals of the study are to: 1) identify the resources potentially at risk, and the societal concerns about the loss of these resources; and 2) identify broad pollution control strategies, and discuss their potential effectiveness and societal effects. In its final report on transported pollutants, OTA will present regionally-oriented descriptions of the potential environmental and societal consequences of these pollutants, and the consequences of actions that might be taken to control them.

1. THE INTERIM DRAFT

This interim draft has been prepared to assist the 97th Congress in its deliberations on the reauthorization of the Clean Air Act. It is the second major synthesis of information prepared during the OTA Assessment of the Impacts of Atmospheric Alterations. The first synthesis, a "staff draft" completed October 1981, was a compendium of information available as of that

date to evaluate long-range transport air pollution. Considerable progress has been made towards completion of the assessment since that preliminary report was made available to the Congress. This current interim product presents results of work designed and commissioned by OTA to further characterize the regional nature of transported pollutants.

The completed assessment will include nearly twenty topics covering a wide range of scientific, technical, economic, and legal concerns. The results presented in this interim draft are from those research areas which have been substantially completed, and for which findings have received professional review. Table 1 outlines the major research activities undertaken in support of the assessment, and indicates the investigations for which findings are presented in this report. Findings from completed investigations are presented in two forms: 1) a brief summary outlining major issues and findings (Volume 1), and 2) excerpts from completed research papers, which present background information, study methodologies, and results in greater detail (bound separately as Volume 2).

This interim draft is comprised of four major areas of investigation: Emissions and the Cost of Control; Resource Effects; Atmospherics; and Legal Mechanisms for Control of Air Pollution. These topics address what is known about: 1) the costs and consequences of controlling long-range transport air pollutants; 2) the effects that are either hypothesized or known to be occurring to resources of concern and to people who use these resources; 3) the relationship between pollutant emissions and the deposition of transported byproducts of these pollutants (i.e., acid deposition and ozone); and 4) the legal mechanisms currently available for controlling transported pollutants.

A. CURRENT EMISSIONS

Table A-1 shows sulfur dioxide and nitrogen oxides emissions inventories for the 31 eastern states.

For 1980, sulfur dioxide emissions in the 31-state region were estimated to be approximately 22.4 million tons. Figure A-1 shows the percent of SO_2 emissions by source category. Utility sulfur dioxide emissions were estimated to be about 16.5 million tons, or abut 75% of the total. Non-utility emissions (including industrial combustion, residential and commercial combustion, and industrial process emissions from refineries, iron and steel producers, and smelters, among others) comprised the remaining 6.4 million tons, or 25% of the total.

For 1980, nitrogen oxide emissions are estimated to be about 14 million in the 31-eastern states. Figure A-2 shows these emissions broken down by sector. Of this total, about 6.2 million tons were from highway vehicles, 4.8 million tons from electric utilities, 2.4 million tons from industrial combustion, and the remainder from many other miscellaneous sources.

TABLE A-1

1980 TOTAL EMISSIONS, OF SO_2 AND NO_x
IN THE 31-EASTERN STATES

	NO_x	SO_2
Alabama	451	759
Arkansas	217	102
Connecticut	134	72
Delaware	52	109
District of Columbia	22	15
Florida	648	1095
Georgia	494	840
Illinois	1006	1471
Indiana	773	2008
Iowa	321	329
Kentucky	531	1121
Louisiana	929	304
Maine	59	95
Maryland	248	338
Massachusetts	254	345
Michigan	690	907
Minnesota	374	260
Mississippi	285	285
Missouri	568	1301
New Hampshire	56	93
New Jersey	406	279
New York	680	945
North Carolina	536	602
Ohio	1145	2647
Pennsylvania	1038	2023
Rhode Island	37	15
South Carolina	260	326
Tennessee	517	1077
Vermont	25	7
Virginia	405	361
West Virginia	452	1088
Wisconsin	421	1138
Region Total	13,992	22,356

SOURCE: U.S.-Canada Work Group 3B under the Memorandum of Intent on Transboundary Air Pollution, personal communication, June 1982.

The remainder of this introductory section provides an overview of long-range transport phenomena, issues associated with transported pollutants, and the conceptual framework used to guide the OTA assessment. In Section II, staff summaries are provided for all twelve completed investigations -- these are designated by a single letter (A through M). The final section of the report (bound separately as appendices) comprises excerpts from completed and reviewed contractor reports.

OTA's final report, which will be completed later this year, will update several of the research areas presented here, and include the results of seven additional study areas, as indicated in Table 1. It will present, insofar as is possible, detailed regional maps and inventories of: likely sources of primary pollutants, secondary pollutant loadings, and crops, materials, forests, soils, lakes, streams, and populations potentially at risk. The final report will further characterize the regional and sectoral distributions of a broad range of costs and benefits associated with transported pollutants, and discuss policy options for Congressional consideration.

TABLE 1

	INTERIM REPORT	FINAL REPORT
I. EMISSIONS AND THE COSTS OF CONTROL		
A. Current Emissions	Included	To Be Updated
B. Stationary Source Controls		
1. Utilities		
a. Costs and Allocation of SO_2 Reductions	Included	
b. Control Technologies for Reducing SO_2 and NO_x Emissions	Included	
c. Effects of Controls on Utility Finances		To Be Added
d. Effects of Controls on the U.S. Coal Market	Included	
2. Other Industrial Sources		To Be Added
C. Mobile Source Control of NO_x	Included	
II. RESOURCE EFFECTS		
A. Agriculture		
1. Effects of Ozone	Included	
2. Sensitivity to Acid Deposition	Included	
B. Forests		
1. Sensitivity to Acid Deposition	Included	
2. Sensitivity to Ozone	Included	
C. Aquatic Resources		
1. Lakes and Streams at Risk	Included	
2. Biotic Effects	Included	To Be Expanded

D. Health Effects		
1. Sulfur Oxides	Included	
2. Nitrogen Oxides	Included	
3. Indirect Effects	Included	
E. Materials		To Be Added
F. Visibility		To Be Added
G. Economic Sectors Involving Resources Considered Sensitive to the Effects of Transported Pollutants	Included	To Be Expanded
III. ATMOSPHERICS		
A. Source-Receptor Relationships	Included	To Be Updated
B. The Chemistry of Acid Deposition		To Be Added
IV. LEGAL MECHANISMS FOR CONTROL OF LONG-RANGE TRANSPORT AIR POLLUTANTS		
A. Interstate Provisions of the Present Clean Air Act	Included	
B. Mechanisms for Control of Transboundary Pollution	Included	
V. POLICY OPTIONS		To Be Added

B. THE ISSUES

1. THE POLLUTANTS OF CONCERN

Air pollutants travel varying distances -- in some cases, hundreds or even thousands of kilometers. For purposes of analysis, transport of air pollutants can be categorized as occurring over three distances:

- Local - up to a distance of 50 km (about 30 miles) from a source;
- Medium range - distances of about 50 km to 300 km; and
- Long range - distances of about 300 km to 2000 km.

Measured pollutant levels are the cumulative result of these local, medium, and long-range effects.

Of those pollutants transported long distances, acid deposition (commonly referred to as "acid rain") and ozone are the focus of this analysis. Acid deposition and ozone result from the chemical transformation of three pollutants: sulfur dioxide, nitrogen oxides, and hydrocarbons. These long-range transport air pollutants (hereafter referred to as "LRTAPs") have been associated with, and in some cases have been identified as a cause of, damage to lakes and streams, forests and soils, agricultural crops, materials, and human health.

"Acid rain" is a term commonly used to refer to a variety of poliutants deposited in both wet and dry form.* Acid deposition -- the term that more

---

* Other acids may contribute to acid deposition, e.g., chlorine (a precursor of hydrochloric acid) may be emitted directly from coal-fired power plants and is usually found relatively short distances downwind from the source. Other potentially injurious substances in deposition

accurately represents the scope of the phenomena -- results from gaseous sulfur and nitrogen oxides, sulfates and nitrates (transformation products of these gases) and from other chemicals in the atmosphere. Acid deposition is a combination of directly-emitted or "primary" pollutants and transformed or "secondary" pollutants. Figure 1 represents the processes involved in acid deposition.

The secondary pollutants -- sulfates, nitrates, and ozone -- can also manifest themselves in other ways. Reductions in visibility are correlated with concentrations of airborne sulfate and nitrate particles. Sulfate particles in the atmosphere, small enough to be deeply inhaled, are of concern for their possible effects on human health. Ozone formed in the atmosphere from nitrogen oxides and hydrocarbons is toxic to plant life and of concern to human health.

Secondary air pollutants have several factors in common: 1) they can form over periods of hours to days and travel hundreds to possibly thousands of kilometers; 2) they cannot be controlled directly, but only by controlling the primary pollutants from which they are formed (or intermediaries that determine their rate of transformation); 3) different secondary pollutants result from the same primary pollutants -- for example, nitrogen oxides can react to form both nitrates and ozone; and 4) they manifest themselves in several ways -- for example, sulfate contributes to both acid rain and visibility reductions.

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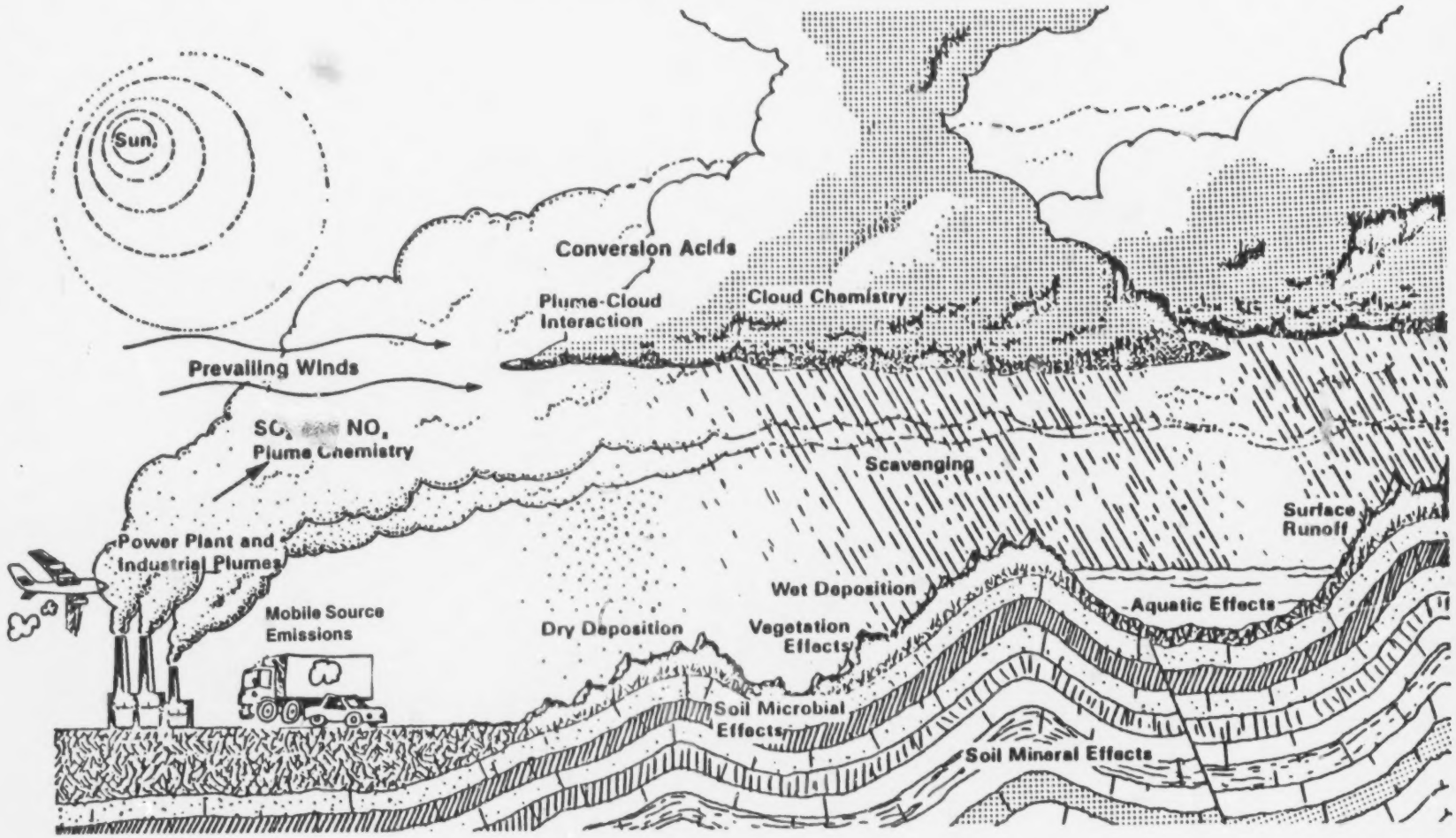
include not only acidic substances but also organic substances (e.g., polycyclic aromatic hydrocarbons), or metals (such as lead, zinc, copper, iron, aluminum, mercury, nickel, etc.). In addition, alkaline substances like soil dust and ammonia may be transported some distance from their origin and may serve to modify the acidity.

Figure 1
Processes and Environmental Effects of Acid Deposition



Source: Adapted from The Acid Precipitation Problem (Corvallis, Oregon: U.S. Environmental Protection Agency, Environmental Research Laboratory, 1976).

2. Overview

Pollutants are emitted into the air, transported by prevailing winds, and, during the process, chemically transformed into other chemical substances which may become acid rain and ozone. Good information exists about the sources and amounts of the primary pollutants -- sulfur dioxide and nitrogen oxides. EPA researchers estimate that between 1950 and 1970 man-made emissions of sulfur and nitrogen oxides in the eastern U.S. have increased significantly. However, sulfur oxide emissions have declined since 1970, while nitrogen oxides have continued to rise.

The sequence of events after primary pollutants are emitted is less well-known. Data about the transport -- the general path the pollutants follow -- are reliable, but limited in temporal and spatial coverage. The intricacies of the chemical transformations that occur are not well understood, although the end products are. Ozone, a secondary pollutant, is known to be formed by chemical reactions involving nitrogen oxides and hydrocarbons. Sulfur oxides and nitrogen oxides are transformed into secondary pollutants such as sulfates and nitrates. The concentrations of sulfates and nitrates in rainfall are highly correlated with the acidity of rainfall.

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The acidity (pH)* of rainfall, as well as the amount of sulfates, nitrates, and other acid-producing substances in rainfall, can be measured fairly well. However, "dry-fall" -- acid-producing substances in either gaseous or particulate form deposited when it is not raining -- is not easily measured. Historical trends in rainfall acidity in eastern North America are difficult to establish because of the uneven quality and quantity of historical data. There are techniques to monitor ozone, but as with measuring acid precipitation, the data are limited in spatial or temporal coverage, especially in rural areas.

Pollutants are often emitted from tall stacks high into the atmosphere, which facilitates transport over long distances and transformation into secondary pollutants. Eventually, pollutants emitted into the atmosphere must be deposited somewhere. Large source regions can be related to large receptor regions using models and other methods. Given the current state of technology, however, pollutants emitted from a single power plant cannot be traced to deposition in a particular sensitive area hundreds of miles downwind, unless newly-developed special tracer techniques are shown to be reliable.**

The amount of pollution deposited within a given area is only part of the information needed to determine the effect on resources. In addition, the

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* pH is a measure of acidity, or the concentration of hydrogen ions. A pH of less than 7 is acidic; greater than 7 is basic. pH is measured on a logarithmic scale. Each unit decrease in pH corresponds to a ten-fold increase in acidity. Each three-tenths of a unit decrease in pH corresponds to a two-fold increase in acid.

** G. J. Ferber, et al., Demonstration of a Long-range Tracer System Using Perfluorocarbon, NOAA Technical Memorandum ERL/ARL 101, NOAA Air Resources Lab, 74 p., 1981.

sensitivity of resources to acid deposition and ozone must be known. Resource sensitivity varies from resource to resource -- and from specific example to example within each resource category. Some resources are directly affected by the pollution received. For example, crops are affected by exposure to ozone, and some building materials are affected by acid-producing substances deposited on their surfaces.

For other resources, characteristics of the local environment also determine sensitivity. For example, lakes are not directly affected by the acid deposition measured by a sampling device, but rather by the amount of acid-producing substances that travel through the watershed to lakes. Sensitivity in this case is determined by the characteristics of the soil and bedrock in the watershed. Potential effects of acid deposition on forests may depend on soil nutrient changes induced by the chemical additions -- here again the sensitivity of the forest resource depends upon the characteristics of the soil. While some fish species are more tolerant of lake acidity than others and some trees are less affected by soil nutrient changes than others, the mediating effects of watershed and soil processes are important determinants of resource sensitivity of acid deposition.

More is known about the effects of acid deposition on aquatic ecosystems -- and the mechanisms that are involved -- than for any other resource. A large body of evidence points to adverse effects from acid deposition on lakes and streams, and suggests that altered water chemistry will be the first response in sensitive aquatic systems. Of greatest concern are small lakes and streams with watersheds having little capability to neutralize acid. This lack of neutralizing capacity can be due either to the chemical composition

and/or thinness of the soils, or to terrain that is so steep or rocky that rainfall runs over it before acid can be neutralized.

Fish are sensitive to both the acidity of water and to toxic metals such as aluminum, which are released from the watershed under acid conditions. Below pH 5, most fish species are eliminated and major changes in lake ecosystem processes may occur. This may be due to high acidity, toxic concentrations of aluminum, or some combination of the two.

Ozone may account for up to 90% of air-pollution-related damage to crops. How much crop damage is due to transported, rather than locally-produced, ozone is uncertain. Observed effects include both qualitative damage (leaf spotting) and quantitative damage (yield reduction).

Damage to crops from acid rain under natural conditions has not yet been observed, although experiments using simulated acid rain have resulted in quantifiable alterations of crop quality and in yield reductions. The role of pollutants mixes may also be significant, since acid deposition seldom occurs in the absence of other pollutants. In some cases, it appears that the threshold for ozone damage to crops may be lowered by the presence of sulfur or nitrogen oxides in the atmosphere. However, other limited data suggest that rather than being synergistic, pollutant mixtures may be merely additive, or possibly antagonistic (i.e., the presence of one pollutant may mitigate the effect of other pollutants).

When considering the effects of transported air pollutants on forests, both direct damage to trees, as well as effects on the soil in which trees grow, must be considered. Because trees are long-lived species, they may be

sensitive to long-term chronic impacts. Ozone damage to the foliage of many tree species has been observed, but the concentrations of ozone necessary before damage occurs is not well-known.

Acid deposition may have both positive and negative effects on forest soils. Forest soils are often more sensitive than agricultural soils because they are not highly managed. In some areas, where nitrogen and sulfur in the soil are in short supply, the nitrates and sulfates in acid rain might improve forest productivity. On the other hand, there is some evidence suggesting that on sensitive soils, acid deposition may remove other essential nutrients, and release metals that are toxic to plants.

Air pollution can damage a broad range of materials -- including building stone, rubber, zinc, steel, paint, and textiles. Ozone is the pollutant that most affects rubber, while other materials are chiefly affected by sulfur oxides. Humidity plays a key role in material damage. Direct contact of sulfur dioxide and sulfate on moist surfaces (forming sulfuric acid) may exceed damage from relatively more dilute acid rain. However, air pollution is only one of many environmental factors (temperature fluctuations, sunlight, salt, microorganisms) that cause materials damage. Therefore, it is difficult to determine what proportion of damage is due to anthropogenic causes. Moreover, it is difficult to discern how much of the materials damage attributed to air pollutants is caused by LRTAPs, as compared to local pollution.

Health effects from transported pollutants are a topic of great controversy. Particles of greatest concern are the fine particulates (less than 0.2 micrometers in size) which may be inhaled deep into lung passages.

Additionally, since such small particles generally do not settle out of the atmosphere by gravity, they tend to be transported long distances. Airborne sulfate concentrations correlate with mortality; whether sulfates are the causative agent or an indicator of other elements in the sulfate-particulate mix is uncertain. Epidemiological studies attempting to show associations between health effects and outdoor concentrations of nitrogen oxides have so far been inconclusive. However, preliminary evidence suggests that nitrogen oxides may interact with other pollutants to induce respiratory problems.

Drinking water criteria for toxic materials are being exceeded in limited areas of the northeastern U.S. Because acidified water can leach metals and other toxic materials from soils and plumbing, acid deposition has been implicated as a potential contributor to the leaching process. Selected populations may be exposed to chronic low-level doses of these toxic substances (e.g., asbestos, copper, aluminum, lead, mercury, and cadmium).

3. WHY LONG-RANGE TRANSPORT AIR POLLUTANTS ARE AN ISSUE

a. General Concern

Long-range transport air pollutants have become an issue because they have been associated with, and in some cases have been identified as a cause of, damage to lakes and streams, forests and soils, agricultural crops, man-made materials, and human health. Because preliminary, although tentative, evidence suggests that the impact of LRTAPs may be significant, some individuals and groups have called upon the Federal Government to control pollutant emissions more stringently than called for under existing law.

Others, pointing to the uncertainties in our knowledge of the causes and consequences of LRTAPs, are concerned that we may act prematurely. They argue that trying to control emissions without fully understanding the processes involved, effects, and alternative mitigation strategies, may waste money and resources and impose unreasonable costs on industry and the public. Reducing total sulfur dioxide emissions by 25% in the eastern U.S. is estimated to cost from $1 to $2.5 billion annually (1980 dollars). Emissions reductions of 50% are estimated to cost from $4 to $5 billion per year. Efforts to control LRTAPs could also significantly affect the coal industry. Stringent sulfur dioxide emissions controls could reduce the demand for high-sulfur coal, displacing miners from their jobs and affecting communities dependent on coal mining.

In addition to concerns about resource damage and costs of control, LRTAPs may impose significant inequities, as their effects might be experienced by different groups, regions, and even nations than those receiving the benefits of the activities that generate acid rain and ozone. Moreover, groups and regions may bear an unequal share of the costs of LRTAP control.

b. Why Transported Pollutants Are a Federal Concern

Long-range transport air pollutants (LRTAPs) have become an issue for potential federal action for two reasons. First, these pollutants do not respect political boundaries. Because pollutants can be transported across several state borders, complete reliance on state-level abatement programs to

control pollution levels within individual states may be ineffective.* In addition, LRTAPs cross the international boundary into Canada, where they have been associated by some with potentially widespread and significant resource damage. Because the Constitution prohibits states from entering into agreements with foreign nations without the consent of Congress (Article 1, Section 10), any pollution control agreements with Canada must be entered into by the Federal Government. The U.S. and Canada signed a Memorandum of Intent in August, 1980, to establish a bilateral research plan to investigate transboundary air pollution and to work toward reaching a bilateral accord on transboundary air pollution. Treaty negotiations began on June 23, 1981 and are still in progress.

Second, existing federal air pollution control mechanisms are governed primarily by the Clean Air Act, which is now being considered for reauthorization by the Congress. Control strategies developed under the Act have focused on control of ambient air concentrations from local sources. The effectiveness of this approach for controlling LRTAPs has been questioned by a number of observers.

3. DECISION-MAKING IN THE FACE OF UNCERTAINTY

Scientific uncertainty surrounding many aspects of the problem complicates the decision of whether, or when, to control transported

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* The shortcomings of state-oriented pollution abatement programs in addressing interstate air pollution does not, however, necessitate federal intervention. States could, in principle, collaborate and negotiate among themselves to form multi-state pollution abatement policies. See Section 102 of the Clean Air Act and Article 1, Section 10 of the U.S. Constitution.

pollutants. Current understanding of transported pollutants allows a description of potential resource damage -- regions of the U.S. most susceptible to damage -- but not a precise specification. The costs of likely control strategies, and who might bear these costs, can be described, but the effectiveness of these control strategies for avoiding resource damage cannot be calculated with confidence. Given the difficulty of quantifying the relationship between emissions reductions and the prevention of resource damage, policy decisions may have to be based on the risks of resource damage, the risks of unwarranted control expenditures, and the distribution of these risks among different groups and regions of the country.

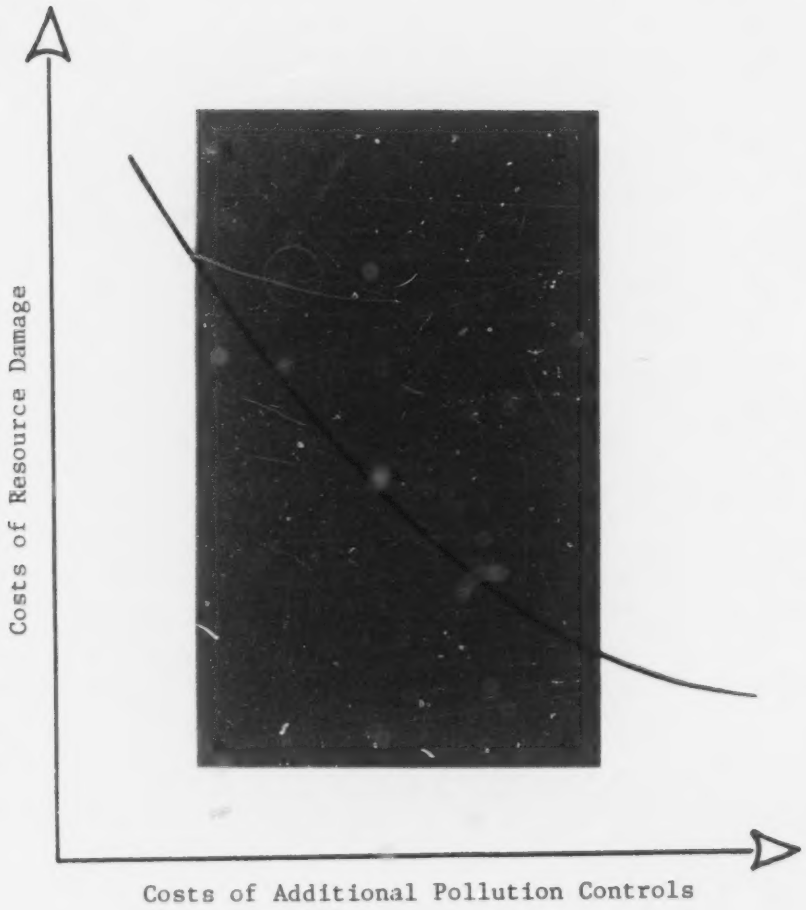
This section briefly discusses two alternative conceptual frameworks for decision-making to illustrate the dimensions of the LRTAP policy issue. The first, benefit/cost analysis, is most helpful for considering the desired extent of pollution control, but is only appropriate when the problem is well understood. The second approach presented is most useful for considering whether to undertake pollution control when the uncertainty surrounding the problem is great.

The curve shown in Figure 2 represents a hypothetical, well-understood relationship between pollution control expenditures and potential resource damage. Each point on this curve represents a given amount of money spent on pollution control, and the resulting amount of resource damage prevented. Benefit/cost analysis considers expenditures to be worthwhile until a point is reached where additional control costs exceed benefits from the resource damage prevented.

Though the benefit/cost test is useful for considering many policy

Figure 2

Alternative Frameworks for Considering LRTAP



questions, the assumptions underlying the analysis and the attributes of the LRTAP problem are poorly matched. To use benefit/cost analysis, the relationship between emissions and resource damage would have to be well-defined. This is not the case. In addition, the test is most effective when costs and benefits have monetary or "market" values. For LRTAPs, many of the resources of concern are not easily assigned monetary values. Finally, the test does not consider the distribution of costs and benefits. As the name long-range transport suggests, the regions that emit pollutants and the regions that receive pollution can be widely separated.

The second conceptual framework is illustrated by the shaded box also shown in Figure 2. The "scenarios" corresponding to the four corners of the box illustrate the complete range of possible outcomes: the effects of the range of available control choices (emission control costs, mitigation costs, etc.), and the range of resource effects (potential damage to lakes and streams, forests and soils, materials, and human health).

Consider first the left-hand side of the diagram -- little or no pollution control expenditures beyond those called for under existing regulations. The range of outcomes, characterized by corners "a" and "b", represents the uncertainties involved in the magnitude of resource damage associated with a given level of LRTAPs.

The right-hand side of the diagram, corners "c" and "d", shows a similar range of outcomes associated with high pollution control expenditures. That this range is similar to the left-hand side of the diagram illustrates the uncertainty about the effectiveness of a chosen control strategy.

Not all points within the shaded box represent equally likely outcomes. Resource damage is more likely to occur with current or increased levels of emissions than with reduced levels; but again, how much damage is currently occurring, or how effective pollution control programs might be in preventing potential damage, is difficult to estimate. By describing the corners of the shaded box, the range of potential outcomes can be bounded, and thus compared. The OTA assessment uses the conceptual framework just described, because it more accurately reflects the attributes of the LRTAP problem.

OTA's analysis characterizes the range of possible scenarios within the shaded area. The first chapter of this interim draft describes one dimension of the issue -- the direct costs of emissions reductions, the distributional implications of alternative legislative proposals currently before the Congress, and the costs ensuing from secondary effects. The second chapter characterizes the natural resource dimension of the issue by estimating the extent of current damage -- and the risks of future damage -- from transported air pollutants.

The third chapter uses state-of-the-art tools to examine the linkage between the sources and receptors of one pollutant of concern, sulfur oxides. Though definitive answers are not available at this time, a plausible description of the current regional interrelationships, and a preliminary examination of the effectiveness of control measures, is presented.

Throughout the report special attention is paid to the regional distribution of risks -- both to those who may bear potentially unnecessary control costs, as well as to those who face potential resource damage. The last chapter examines the current mechanisms under the Clean Air Act available

to those involved in interregional air pollution disputes -- both interstate and international.

Policy options are not included in this interim draft, but will be presented in the final report.

FIGURE A-1

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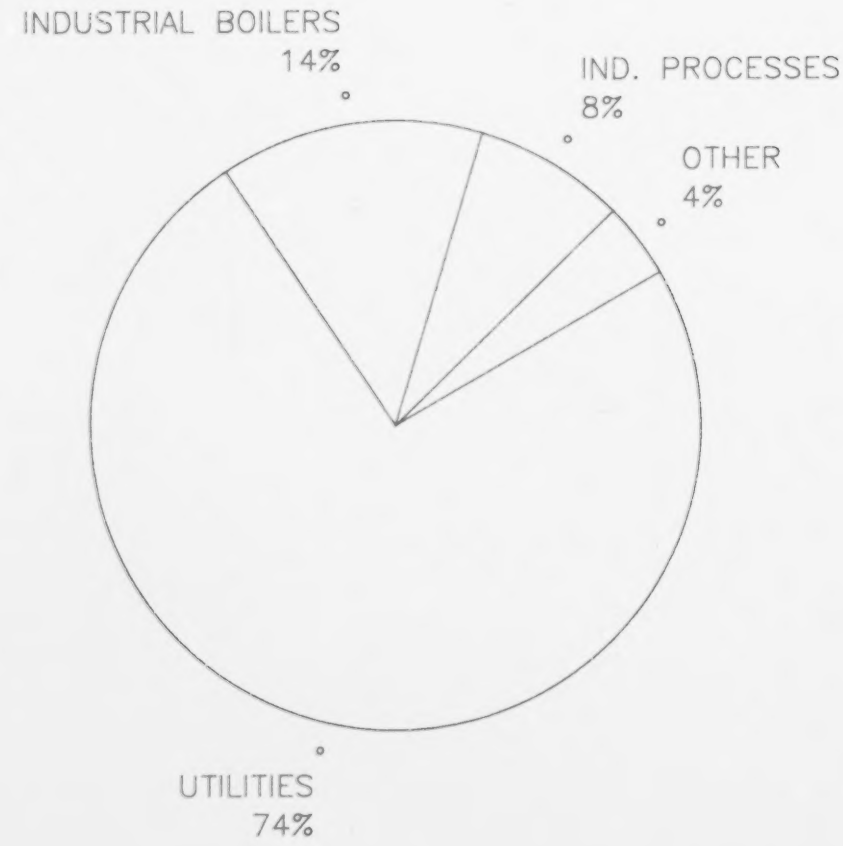
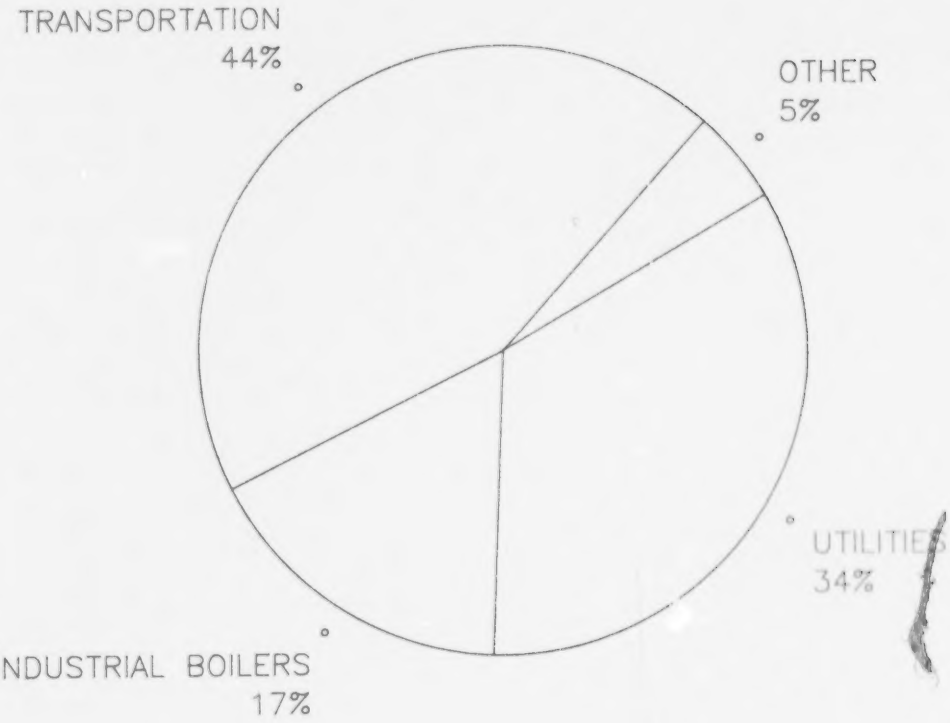
SULFUR DIOXIDE EMISSIONS IN THE 31-EASTERN STATES
PERCENT OF 1980 EMISSIONS BY SOURCE CATEGORIES



SOURCE: U.S.-Canada Work Group 3B under the Memorandum of Intent on Transboundary Air Pollution, personal communication, June 1982.

FIGURE A-2

NITROGEN OXIDES EMISSIONS IN THE 31-EASTERN STATES
PERCENT OF 1980 EMISSIONS BY SOURCE CATEGORY



SOURCE: U.S.-Canada Work Group 3B under the Memorandum of Intent on Transboundary Air Pollution, personal communication, June 1982.

B. ALLOCATION OF SULFUR DIOXIDE EMISSIONS REDUCTIONS AND THE COSTS OF CONTROL

Summary

Decisions regarding long-range transport air pollutant (LRTAP) control require knowledge of the costs and distributional consequences of various control strategies. Both the issues of the amount of emissions to be reduced, and the manner in which reductions are to be achieved, involve questions of cost. For a given method of achieving reductions, the greater the reduction, the greater the cost. Alternatively, given the same emissions reduction target, different implementation strategies may entail different costs, i.e., one strategy may be more cost-effective than another.

In addition to questions of cost, alternative reduction implementation strategies may have different distributional consequences. Depending on the design of the strategy, one region or state or economic sector may bear a greater share of the emissions reduction burden than another. This section examines the costs and distributional consequences of various emissions reduction strategies.

Following the design of most of the acid rain control proposals to date, emissions reductions in the eastern 31-state region, rather than the entire U.S., will be considered. Due to analytical limitations, only the costs of sulfur dioxide emissions reductions from utilities are presented. No estimates of the costs of utility nitrogen oxide emissions reductions, nor costs of industrial-sector emissions reductions, are included.

1. SULFUR DIOXIDE EMISSIONS IN 1980

Sulfur dioxide emissions for 1980 in the 31 eastern states are estimated

to be about 22.5 million tons. Approximately 70 percent of this total, or 16 million tons, originates from fossil fuel combustion by electric utilities.* Under current regulations, EPA-approved State Implementation Plans (SIPs) require utilities to reduce these emissions by approximately 1.5 million tons, to 14.5 million tons per year.

To aid in evaluating proposals to reduce the long-range transport of air pollutants, OTA has developed estimates of the cost to utilities in the 31-state region of reducing sulfur dioxide emissions below the 14.5 million ton SIP-compliance level. The model used in generating these estimates is described briefly in Appendix BB.**

II. THE COSTS OF VARIOUS LEVELS OF EMISSIONS REDUCTIONS

To illustrate how the costs of controlling sulfur dioxide emissions may be influenced by the amount of emissions reductions required, OTA has estimated the costs of achieving a series of maximum emissions limits (termed "caps") ranging from 1.0 to 4.0 pounds of sulfur dioxide emitted per million Btu of energy input.*** Figure B-1 displays the results of this analysis. Emissions reductions range from 2.2 to 10.3 million tons per year. Although these emissions reductions include the reductions required under current law,

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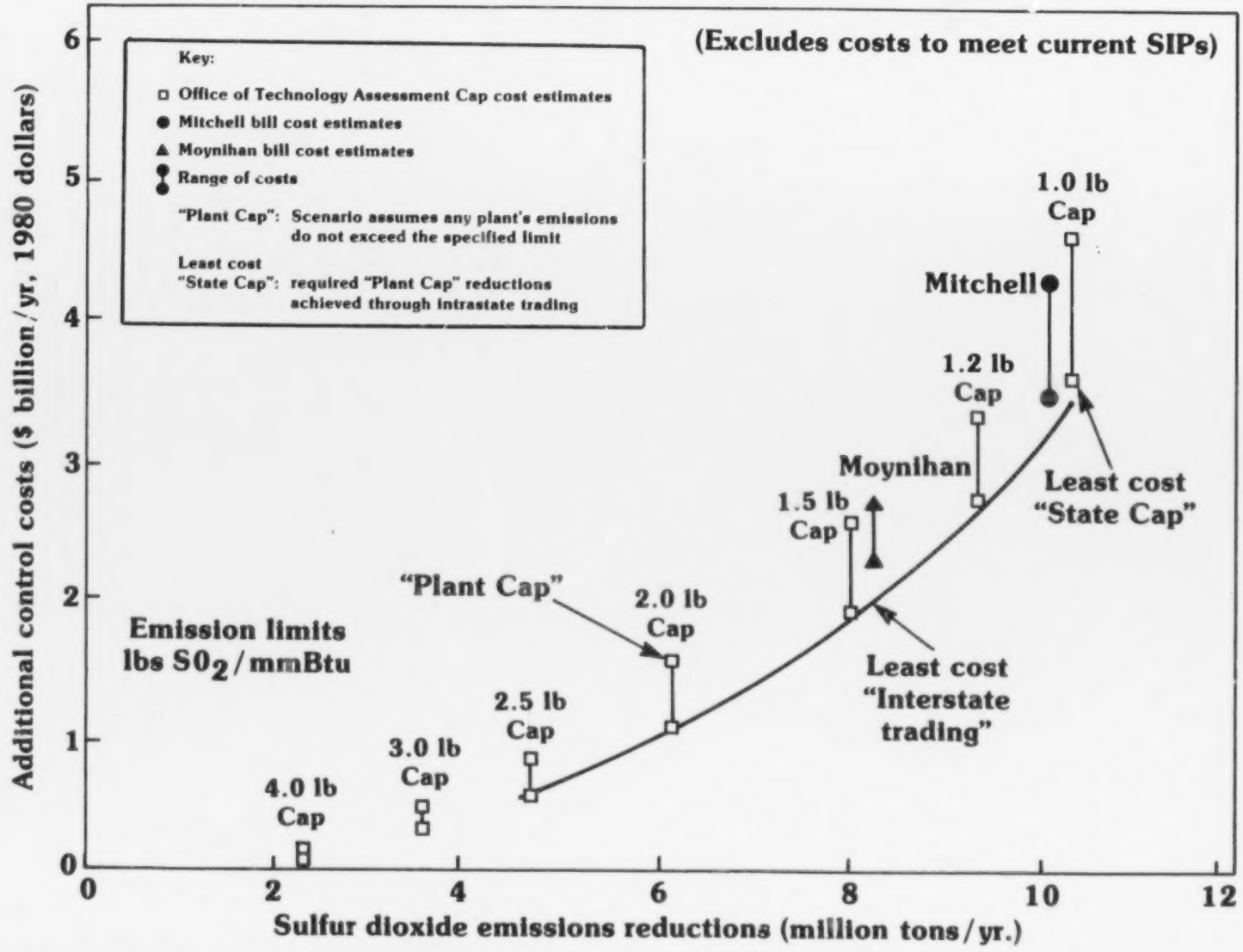
* The slight inconsistencies between the emissions data presented here and in Section A of the report are due to the use of different emissions inventories. Estimates of utility emissions regionwide should be considered accurate to within 5-10%.

** The model used for this analysis is the AIRTEST model, run by E.H. Pechan and Associates, Inc. AIRTEST was modified from a larger model used in several earlier major assessments, including the New Source Performance Standards (NSPS) review, the Ohio River Basin Energy Study (ORBES), and the Acid Rain Mitigation Study (ARMS).

*** These "emission rates" reflect the amount of SO_2 produced per unit of energy in the fuel burned.

FIGURE B-1

Cost of Sulfur Dioxide Control



SOURCE: Office of Technology Assessment, 1982.

the costs presented consider only those reductions that would be required beyond SIP-compliance.

A line joining two symbols indicates the range of costs which might be associated with the specified emissions reduction. The upper estimate assumes that each state would achieve the required reductions by imposing uniform emissions ceilings on all utility plants (labeled "plant cap"). The lower estimate assumes that each state will choose the most cost-effective way of meeting the required emissions reductions (labeled "state cap"). This could be achieved by some policy that would allow emitters within the state to trade their emissions reduction obligations, or by state-level requirements for emissions reductions from those plants that can reduce emissions least expensively. However, no interstate trading occurs. These cost estimates are presented in in tabular form in Table B-1.

The curve on the graph represents the "regional least-cost" approach. This strategy assumes a "perfect market" with total freedom to trade emissions reduction obligations across the entire region. For emissions reductions less than 2.5 million tons below SIP compliance, e.g., in the 3.0 to 4.0 pound cap area of the graph, the "regional least cost" is not significantly different from estimated costs of a "state cap".

As expected, costs increase as emissions reduction requirements increase -- from less than $1 billion annually for reductions less than 6 million tons per year, to between $4 and $5 billion for a reduction of 10.3 million tons. These are first-year, annualized costs in 1980 dollars.* Moreover, the

---

* First year, annualized costs average capital costs evenly over each year of the life of the investment. Because fuel and operation and maintenance costs can vary from year to year (i.e., real energy and labor

TABLE B-1

THE COSTS OF REDUCING SULFUR DIOXIDE EMISSIONS
IN THE 31 EASTERN STATES

(Excludes costs to meet current SIPs)

Emissions Caps (lbs SO_2 mm Btu)	Reduction below 1980 Emissions (millions of ton/year)	Reduction Below SIP Emissions (millions of tons/year)	"State Cap" Cost*		"Plant Cap" Cost*	
			$ billions per year	$/ton	$ billions per year	$/ton
4.0	2.3	0.9	0.1	90	0.1	160
3.0	3.5	2.1	0.3	150	0.5	230
2.5	4.7	3.2	0.6	180	0.9	270
2.0	6.1	4.7	1.1	240	1.6	350
1.5	8.0	6.6	2.0	310	2.7	410
1.2	9.3	7.9	2.9	370	3.5	450
1.0	10.3	8.8	3.8	430	4.9	550

* All costs are in 1980 dollars. "State least cost" assumes that each state will choose the most cost-effective way of meeting the required emissions reductions. "Plant cap" assumes that each state would achieve its allocated reductions by imposing a uniform emissions ceiling on each utility plant. Average costs are calculated on the basis of emissions reductions below SIP-compliance levels.

Source: Office of Technology Assessment, based on AIRTEST analysis by E.H. Pechan & Associates.

marginal costs of control increase when larger emissions rollbacks are required. That is, the cost of increasing an emissions reduction by 1 million tons is greater for an increase from 7 to 8 million tons (approximately $550 million) than for an equal increase from 5 to 6 million tons (approximately $300 million).

For comparison, estimates of the costs of two legislative proposals have been included on the graph. S.1709 (the Moynihan bill) would decrease sulfur dioxide emissions by approximately 8 million tons, and cost between $2.4 and $2.9 billion annually. S.1706 (the Mitchell bill) would decrease annual emissions by 10 million tons, and cost between $3.7 and $4.5 billion per year.

Figure B-2 compares OTA's estimates to estimates of the costs of acid rain control measures made by other groups. Only the emissions reductions for which cost estimates were calculated are included on the graph. Because the AIRTEST model does not calculate the cost of achieving SIP compliance, OTA's estimates are presented as the costs of reductions beyond SIP compliance. The PEDCo study was prepared for DOE, the Teknekron analysis was prepared for EPA and DOE, and the ICF analyses were performed for EPA, DOE, Edison Electric Institute, and the National Wildlife Federation. The Peabody, PEDCo, and DOE estimates are reductions below actual 1980 emissions, and the ICF and Teknekron model estimates are displayed as emissions reductions below projected 1990 emissions levels. All costs are in 1980 dollars. Emissions reductions and costs are presented in tabular form in Table B-2.

In the emissions reductions range of 6 to 9 million tons per year, OTA's

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costs might either increase or decrease over the next two decades), current fuel and operation and maintenance costs (rather than an average based on assumed trends) are added to capital costs to calculate yearly total control costs.

FIGURE B-2

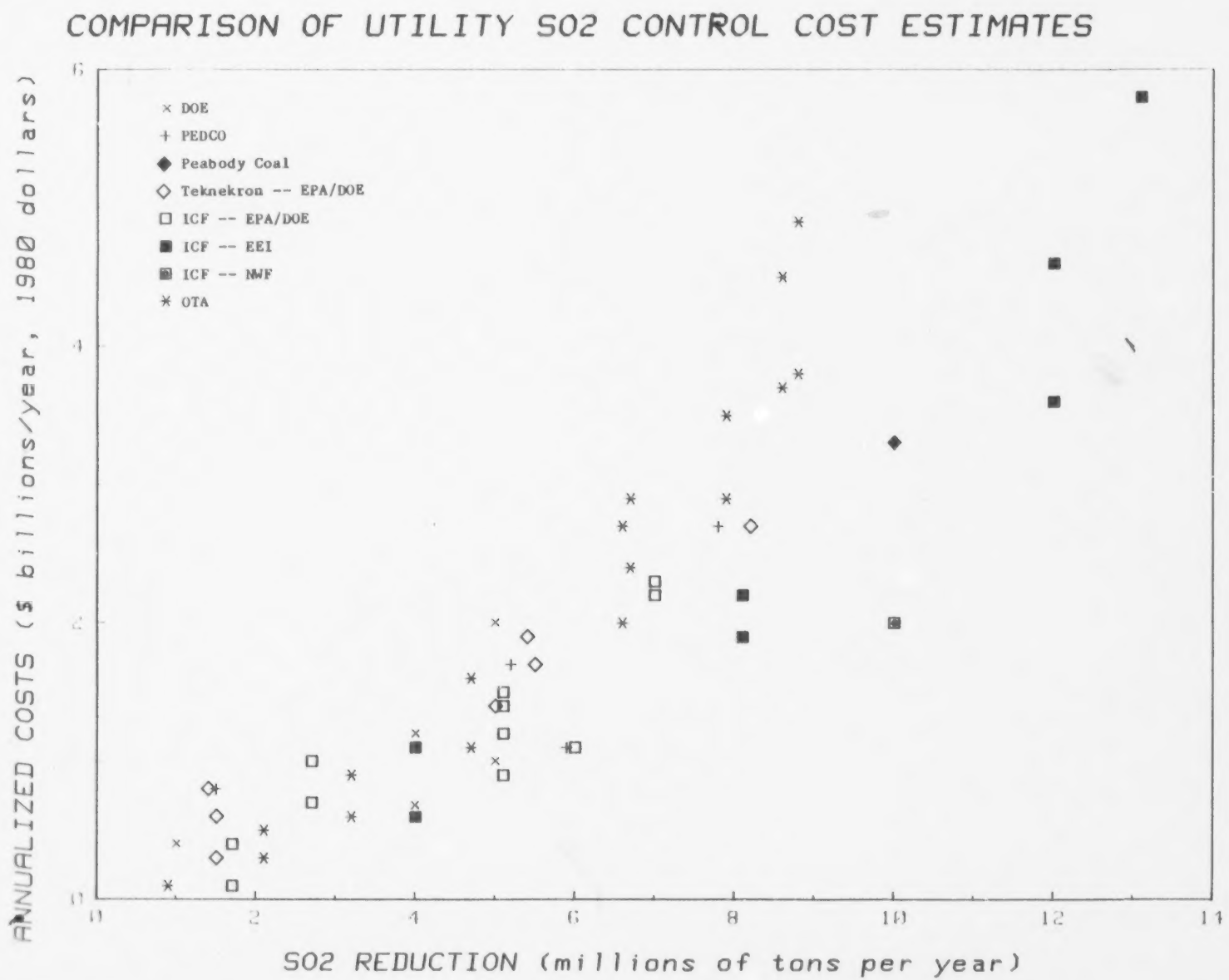
COMPARISON OF UTILITY SO2 CONTROL COST ESTIMATES
ANNUALIZED COSTS ($ billions/year, 1980 dollars)
SO2 REDUCTION (millions of tons per year)
DOE
PEDCO
Peabody Coal
Teknekron -- EPA/DOE
ICF -- EPA/DOE
ICF -- EEI
ICF -- NWF
OTA
0
2
4
6
8
10
12
14

TABLE B-2 COMPARISON OF UTILITY SO_2 CONTROL COST ESTIMATES

Source	SO_2 Reduction (millions of tons per year)	Annualized Costs ($ billions per year, 1980 dollars)
Reductions Below 1980 Emissions		
DOE[1]		
Coal-switching	4.0	0.6-1.2
FGD	5.0	1.0-2.0
Coal Cleaning	1.0	0.3-0.4
PEDCO[1]		
Coal-Switching	5.9	1.1
FGD 60%	5.2	1.7
FGD 90%	7.8	2.7
Coal Cleaning	1.5	0.8
Peabody Coal[3]	10.0	3.3
Reductions Below Projected 1990 Emissions		
Teknekron -- EPA/DOE[1]		
4 lb. Cap	1.5	0.3
4 lb. Cap + Local Coal	1.5	0.6
2 lb. Cap	5.5	1.7
2 lb. Cap + Local Coal	5.4	1.9
30% FGD	5.0	1.4
50% FGD	8.2	2.7
Coal Cleaning	1.4	0.8
ICF -- EPA/DOE[1]		
10% Regional	1.7	0.1
10% Each State	1.7	0.4
4 lb. Cap	2.7	0.7
4 lb. Cap + Local Coal	2.7	1.0
15¢/lb. Tax	6.0	1.1
30% Regional	5.1	0.9
30% Regional + Local Coal	5.1	1.2
30% Each State	5.1	1.4
30% Each State + Local Coal	5.1	1.5
2 lb. Cap	7.0	2.2
2 lb. Cap + Local Coal	7.0	2.3
ICF -- EEI[2]		
S.1706-offset growth	13.1	5.8
S.1706-offset growth, NO_x & ind.trades	12.0	3.6-4.6
S.1706-SIP credit, NO_x & ind.trades	8.1	1.9-2.2
EEI Rollback	4.0	0.6-1.1
ICF -- NWF[2]	10.0	2.0
Reductions Below 1980 SIP Emissions		
Office of Technology Assessment[4]		
1.0 lb. Cap	8.8	3.8-4.9
1.2 lb. Cap	7.9	2.9-3.5
1.5 lb. Cap	6.6	2.0-2.7
2.0 lb. Cap	4.7	1.1-1.6
2.5 lb. Cap	3.2	0.6-0.9
3.0 lb. Cap	2.1	0.3-0.5
4.0 lb. Cap	0.9	0.1
Mitchell	8.6	3.7-4.5
Moynihan	6.7	2.4-2.9

TABLE B-2 (CONT)

Footnotes to Comparison of Utility SO_2 Control Cost Estimates

1. Department of Energy, Costs to Reduce Sulfur Dioxide Emissions, DOE/PE-0042, March, 1982.

2. ICF, Inc., "Summary of Acid Rain Analyses Undertaken by ICF for the Edison Electric Institute, National Wildlife Federation, and Environmental Protection Agency", prepared for Edison Electric Institute, May, 1982.

3. Chris Farrand, Peabody Coal Company, Testimony Before the Senate Environment and Public Works Committee, October 29, 1981.

4. Office of Technology Assessment, Staff Briefing Memo for the Senate Environment and Public Works Committee, June 2, 1982, (Revised June 4, 1982).

cost estimates are in general higher than the other model estimates. For emissions reductions of approximately 5 million tons, the various annual cost estimates range from $0.9 to $2.0 billion per year. For reductions of 8 million tons (50% below current utility emissions), the range increases from $1.9 to $3.5 billion annually. The largest emissions reduction calculated -- a 13 million ton reduction below projected 1990 levels analyzed by ICF for EEI -- is estimated to cost approximately $6 billion per year.

Appendix BB presents estimates of the cost impacts of S. 1706 (the Mitchell bill) performed by specific utilities.

III. STATE-BY-STATE EMISSIONS REDUCTIONS AND COSTS OF CONTROL STRATEGIES

Thus far, costs of control have been presented at the aggregate, regional level. Costs of control will vary considerably from state to state, depending upon each state's emissions reduction requirements, and the costs of available emissions reductions in each state. In addition, regional costs and their distribution vary according to the specific control strategy chosen.

OTA has analyzed the costs and distributional consequences of a number of approaches to allocating an 8 million ton reduction in sulfur dioxide emissions to states. A detailed analysis of alternative allocation formulae is presented in Appendix BB. The results of the analysis of 7 of these alternative approaches to allocation are presented here:

(1) 50% Emissions Reductions in Each State: Each state is required to reduce emissions by an equal percentage, regardless of the relative costs of emissions control or the stringency of the state's existing emissions regulations.

(2) S. 1709 -- The Moynihan Bill: This "targetted retrofit" approach requires that those plants emitting greater than 50,000 tons of sulfur dioxide per year, at a rate greater than 3 pounds of sulfur dioxide per million Btu, must reduce their emissions by 85% below current

levels.* This formula applies to about 60 plants in 16 states.

(3) Maximum Emissions per Unit of Energy Input (Emissions "Cap"): Emissions have to be reduced so that each plant complies with an emissions limit of 1.5 pounds of sulfur dioxide per million Btus of energy input.

(4) Average Emissions per Unit of Energy Input: Emissions have to be reduced so that a state complies with an emissions limit of 1.3 pounds of SO_2 emitted per million Btu of energy input, averaged across all the fossil fuel utility plants in the state. Unlike the cap approach, some plants are allowed to exceed the specified rate as long as there are compensating plants emitting below the rate.

(5) Average Emissions per Unit of Total Electricity Output: This approach gives credit to states that rely on electric-generating facilities that do not emit SO_2 by specifying that each state comply with an average emission rate of 11.3 pounds of SO_2 per MWhr of total electricity output.

(6) Average Emissions per Person: Because this allocation approach does not rely on either inputs to or outputs from the electricity generating process, total (utility, industrial, etc.) emissions can be considered. This approach would require each state to comply with an average emission rate of 200 pounds of total sulfur dioxide emissions per state resident.

(7) Average Emissions per Area: By requiring that each state comply with an average emission rate of 18 tons of SO_2 per square mile annually, this approach would allocate reductions to states according to their emissions density (emission per unit of land area).

The actual state-by-state costs that would be incurrred from a given allocation formula are determined by (1) the amount of reductions allocated to each state, which depends upon the control policy chosen; and (2) the costs of available emissions reduction opportunities to each state, which depend upon the type and number of electric plants in service, and their levels of current

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* In addition, S. 1709 provides that no state is required to reduce utility emissions, calculated after SIP compliance, by more than 50% (if the state's average sulfur dioxide emission rate is less than or equal to 2 pounds per million Btus), or more than 75% (if the state's average emission rate is greater than 2 pounds per million Btus). While reductions are allocated to states on the basis of these "targetted facilities," the bill leaves the actual implementation of reductions to the discretion of the states.

emissions. States that already have relatively low emissions per unit of energy input or electricity output may not have as much opportunity to use less expensive control options as those states with higher emission rates. The latter states may be able to achieve relatively large reductions at lower costs per ton. Appendix BB presents detailed data on emission rates and average costs of control for each state.

Table B-3 presents the 31-state region cost of these 7 alternative approaches to allocating reductions to states. The costs are shown to vary from formula to formula, ranging from a low of $2.0 billion per year ($310 per ton of SO_2 removed) for the lower estimate of the 1.5 pound cap, to a high $3.5 billion per year ($540 per ton of SO_2) for the upper estimate of the emissions density approach.

Table B-4 shows the distribution of state-by-state costs for each allocation approach. Control costs are displayed in terms of ranges of annual costs per kilowatt-hour as a percentage of 1979 electricity prices. Some states are consistently allocated relatively large costs. These states are: Indiana, Kentucky, Missouri, Ohio, Pennsylvania, Tennessee, and West Virginia. For the allocation approaches that result in the widest variation of required emissions reductions -- such as those indicated in S. 1709, "emissions per capita", and "emissions densities" -- the distribution of costs varies a great deal. In these cases, some states are allocated very large costs and others incur no costs at all. States vary as to the degree to which their costs are affected by alternative approaches. For instance, Arkansas and Louisiana are allocated relatively low costs under all of the approaches presented. In other states, such as Massachusetts, Delaware, and Kentucky, control costs are influenced strongly by the allocation approach used.

TABLE B-3

REGIONAL COSTS OF ALTERNATIVE APPROACHES TO ALLOCATING AN 8 MILLION TON REDUCTION IN SO_2 EMISSIONS

Allocation Approach[1]	State Least Cost[2] ($ bil/ year)	Average Cost[3] ($/ton)	Plant Cap[2] ($ bil/ year)	Average Cost[3] ($/ton)
50% Each	2.6	390	3.2	480
S.1709	2.5	370	2.9	440
1.5 lb. Cap	2.0	310	2.7	410
1.3 lb. Average	2.1	320	2.7	400
Total Electricity	2.1	310	2.6	390
Per Capita	2.6	390	2.9	440
Emissions Density	3.4	520	3.5	540

1. Alternative approaches explained in text.

2. All costs in 1980 dollars. The allocation formula is used to allocate emission reductions to states. "State least cost" assumes that each state will choose the most cost-effective way of meeting the required emissions reductions. "Plant cap" assumes that each state would achieve its allocated reductions by imposing uniform emissions ceiling on each utility plant.

3. Average costs are calculated on the basis of emissions reductions below SIP compliance levels.

Source: Office of Technology Assessment, based on AIRTEST analysis.

TABLE B-4

STATE-BY-STATE COSTS OF ALTERNATIVE ALLOCATION FORMULAE

(costs as a percent of 1979 electricity rates)

Percent of 1979 Rates
0% = -
1-3% = +
3-5% = ++
5-8% = +++
8-12% = ++++
12-16% = +++++
16-20% = ++++++

State	50% EACH	MOYNIHAN	1.5 LB CAP	1.3 LB AVE	ELEC, TOTAL	PER CAPITA	PER AREA
Alabama	*	*	*	*	-	**	-
Arkansas	*	-	-	-	-	-	-
Connecticut	-	-	-	-	-	-	-
Delaware	***	-	*	*	*	***	*****
D.C.	-	-	-	-	-	-	-
Florida	**	*	*	*	*	-	-
Georgia	***	***	***	***	***	**	-
Illinois	*	*	*	*	*	*	*
Indiana	***	*****	***	****	****	******	******
Iowa	*	-	*	*	*	*	-
Kentucky	**	******	***	***	***	****	*
Louisiana	**	-	-	-	-	-	-
Maine	*	-	*	*	-	-	-
Maryland	**	-	*	**	*	-	***
Massachusetts	***	-	*	*	*	-	*****
Michigan	*	*	*	*	*	-	-
Minnesota	***	-	*	*	-	-	-
Mississippi	**	**	**	-	*	-	-
Missouri	**	***	***	***	****	***	*
New Hampshire	***	****	**	***	***	*	-
New Jersey	*	-	*	-	-	-	***
New York	*	*	*	-	-	-	*
North Carolina	****	-	*	*	*	*	-
Ohio	**	***	***	***	***	***	*****
Pennsylvania	**	***	**	**	**	***	****
Rhode Island	-	-	-	-	-	**	-
South Carolina	**	-	*	**	-	-	-
Tennessee	*	*****	**	***	***	***	*
Vermont	-	-	-	-	-	-	-
Virginia	***	-	*	*	-	-	-
West Virginia	***	***	***	***	***	*****	****
Wisconsin	*	*	*	**	*	*	-

The AIRTEST model results illustrate that both the regional costs of control and the distributional consequences depend upon the approach chosen for allocating emissions reductions to states. Therefore, if a decision is made to reduce sulfur dioxide emissions in the eastern 31-state region, the choice of allocation policy involves both the political issues of who should bear the burden of reducing emissions as well as economic issues of cost.

THE COSTS OF CONTROL IN PERSPECTIVE

State-by-state costs of control have been presented as percentages of 1979 electricity rates. These percentages reflect not only the estimated costs of acid rain controls, but current electricity prices as well. To place these percent increases in perspective, OTA has ranked states according to (1) their 1979 electricity rates, and (2) 1979 rates plus model-estimated costs of SO_2 reductions.

Figure B-3 displays state-by-state cost as a percentage of electricity rates for S. 1706 -- a 10 million ton reduction in annual sulfur dioxide emissions. Figure B-4 presents the ranking of electricity costs within the region before and after emissions reductions required by S. 1706.* With the exception of Missouri, no state is estimated to change rank by more than four places. Several of the states with the largest emissions reduction

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* Percent rate increases are derived by combining model estimates of the annualized cost of emissions reductions (using 1979 estimated fuel prices and scrubber costs) with 1979 residential electricity price data. These figures are more accurately interpreted as control costs per kwhr of electricity generated, expressed as a percentage of 1979 rates. Because electricity rates are set by government regulatory agencies, the actual impact of emissions reduction costs on rates depends upon the regulatory treatment of pollution control expenditures in such areas as depreciation schedules, tax credits, fuel price adjustment clauses, and construction work in progress.

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FIGURE B-3

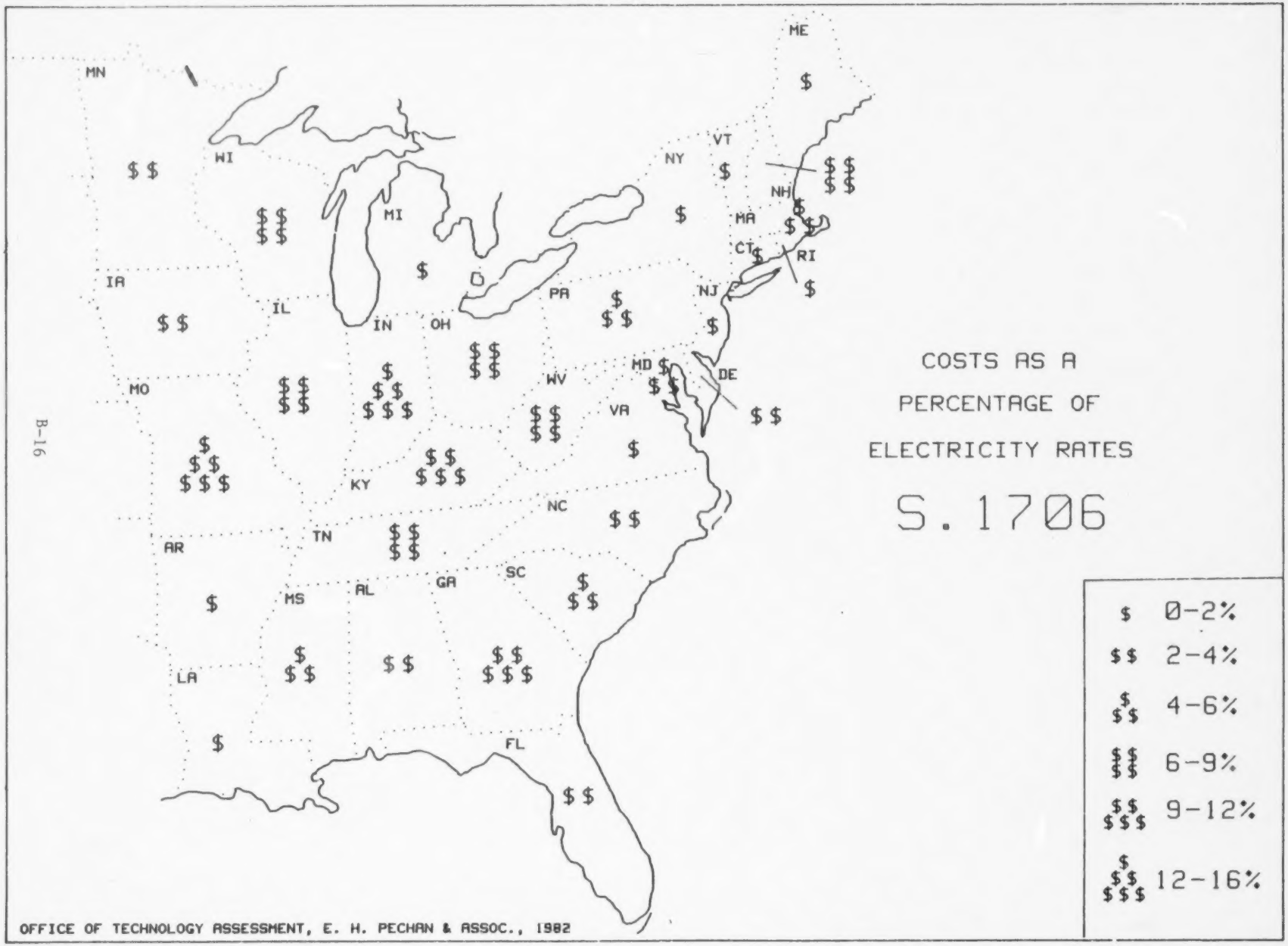
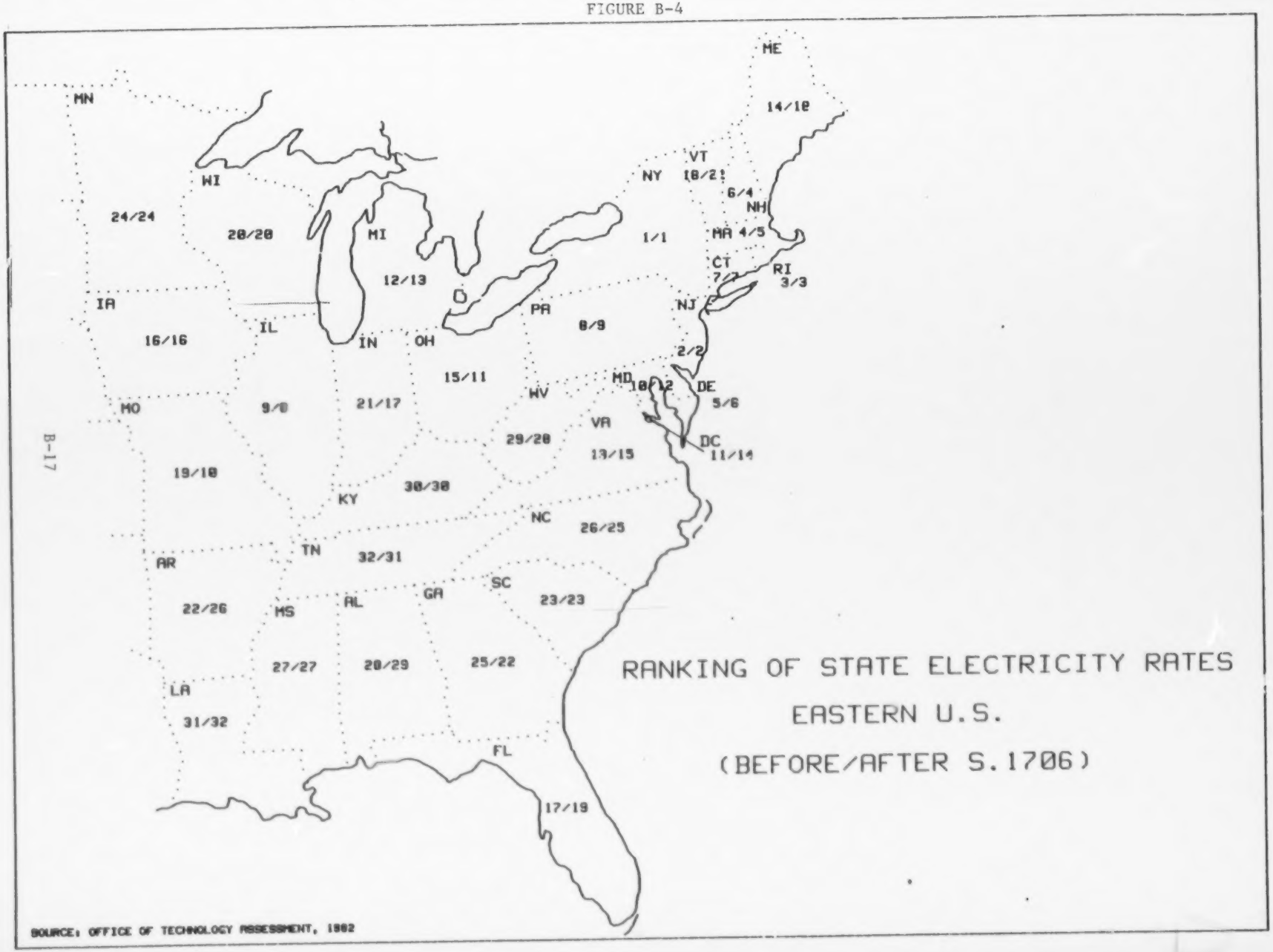

ME
MN
VT
NY
WI
NH
MI
MA
CT
RI
IA
PA
NJ
IL
IN
OH
MD
DE
MO
WV
VA
KY
NC
TN
AR
SC
GA
AL
MS
LA
FL
COSTS AS A
PERCENTAGE OF
ELECTRICITY RATES
S.1706
$ 0-2%
$$ 2-4%
4-6%
6-9%
9-12%
12-16%
OFFICE OF TECHNOLOGY ASSESSMENT, E. H. PECHAN & ASSOC., 1982

FIGURE B-4


ME
14/10
MN
VT
NY 10/2
6/4
NH
WI
24/24
20/20
MI
1/1
MA 4/5
CT
7/7
RI
3/3
12/13
IA
NJ
PA
8/9
16/16
IL
IN
OH
2/2
15/11
MD 10/12
DE
5/6
MO
9/8
21/17
WV
VA
29/28
13/15
DC
11/14
19/10
KY
30/30
NC
26/25
TN
32/31
AR
SC
GA
23/23
22/26
MS
AL
27/27
28/29
25/22
LA
31/32
FL
17/19
RANKING OF STATE ELECTRICITY RATES
EASTERN U.S.
(BEFORE/AFTER S.1706)
SOURCE: OFFICE OF TECHNOLOGY ASSESSMENT, 1982

requirements, such as Tennessee, West Virginia, and Kentucky, presently have relatively low electricity prices, and are not estimated to change rank significantly under S. 1706.

To place the projected rate increases into perspective, Table B-5 shows the change in electricity prices over the past decade. The producer price index for electricity (the price of electricity to industry) increased at an average annual rate of 11.8% between 1970 and 1980. The consumer price index for electricity has increased 9.1% per year over this period. OTA's model projects rate increases for a 10 million ton reduction to vary from 0 to almost 20%.

In 1980, the average household spent approximately $400 on electricity purchases. This amounts to 1.9% of the median family income.** A 10% electricity rate increase would thus amount to about 0.2% of the average household's income; a 50% rate increase would amount to 0.9% of the average household's income. These values are calculated on the basis of national averages; actual values would vary from state to state and from household to household.

V. CONTROL COSTS AND REGULATORY POLICY

In calculating cost increases, OTA's model assumes that each plant will choose the most cost-effective method of emissions reduction. However, state regulatory policies can affect this choice -- and hence the costs -- in ways not treated by the model. For example, most state regulatory commissions have

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** This figure is based on a national monthly average of $33 for a 500 kwh bill. See DOE/EIA-0040 (81), Typical Electric Bills - January 1, 1981, p. 6. Medium family income for 1980 was $21,000. See Statistical Abstract of the United States - 1981.

TABLE B-5

TRENDS IN ELECTRICITY PRICES -- 1970 - 1980

	Consumer Price Index for Electricity	Producer Price Index for Electricity
1970	106.2	105.9
1971	--	--
1972	--	--
1973	124.9	129.3
1974	147.5	163.1
1975	167.0	193.4
1976	177.6	207.6
1977	189.3	232.9
1978	203.4	250.6
1979	219.1	270.2
1980	253.4	321.6
Percent Increase		
1970-1980	138%	203%
Annual	9.1%	11.8%

Source: Statistical Abstract of the United States, 1981.

"automatic fuel adjustment clauses" that allow utilities to pass the costs of increased fuel prices due to fuel-switching to consumers immediately. However, for emissions controls requiring capital investment, such as scrubbers, most states require utilities to wait until the equipment becomes operational before charging ratepayers. This practice may create a bias against capital investment in pollution control equipment in utility management decision-making, and actually increase the lifetime costs of a plant.*

The model used by OTA also assumes that all the costs of control are allocated to the state whose utility owns the facilities that would be required to reduce emissions. The accuracy of this assumption depends on the policy chosen for allocating of costs. To relieve utilities or states that are allocated particularly large emissions reductions, costs could be spread over electricity consumers in other areas.**

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* Some rate-making commissions have begun to allow "construction work in progress" (CWIP) to be included in the rate base to minimize the burden of construction programs on the financial condition of utilities, but this practice is not widely applied at this time. This measure would increase electricity rates to consumers in the short run, but decrease the costs of producing electricity over the lifetime of the plant. A recent study estimates that the increased lifetime costs of not including CWIP in the rate base are about equal to the costs of a scrubber. See Congressional Budget Office, The Clean Air Act, Electric Utilities, and the Coal Market, April 1982.

** For example, a bill proposed by Senator Danforth would spread the costs of control over the entire region specified by the bill by requiring that each utility in the region contribute a fee of 0.3 cents per kwh of electricity sold to an "Acid Deposition Reduction Trust Fund". The money in the trust fund is then to be allocated to those states required to reduce emissions by the bill.

C. CONTROL TECHNOLOGIES FOR REDUCING SULFUR DIOXIDE AND NITROGEN OXIDES EMISSIONS

Summary

Acid deposition and ozone are believed to result primarily from the chemical transformation of three pollutants: oxides of sulfur, oxides of nitrogen, and hydrocarbons. This section discusses the most significant control technologies available or under development for reducing emissions of oxides of sulfur and nitrogen. The combustion of fossil fuel is the major source of these pollutants. Three general approaches are used for controlling fossil-fuel emissions: pre-combustion, during combustion, and post-combustion. All three of these approaches have been successfully utilized to reduce emissions from existing sources.

Control approaches also differ widely in the amount of emissions reductions they are capable of achieving and in their cost-effectiveness. Each technology is most cost-effective in a particular range of emission reductions. For instance, the pre-combustion approach of physical coal cleaning is only technically feasible for sulfur dioxide reductions of less than about 40 percent. On the other hand, the post-combustion approach of flue-gas desulfurization is only cost-effective in the 50 to 90 percent sulfur dioxide removal range. The control technique appropriate for a given facility thus depends a great deal on the level of reduction desired. Emission control techniques also vary considerably in their stage of development -- the techniques discussed below are characterized as either "in use," "available,"

or "emerging."[1]

Controlling Sulfur Dioxide Emissions

I. Precombustion Approaches

Sulfur dioxide is formed when sulfur, an element naturally present in fuel, is oxidized during the combustion process. The greater the concentration of sulfur in the fuel being burned, the greater the production of sulfur dioxide gas. One way to reduce SO_2 emissions is to use fuels with lower concentrations of sulfur. Two important problems with fuel-switching are: 1) low-sulfur coal is more expensive than high-sulfur coal, especially in the east and midwest, and 2) some of the fuel-handling facilities, boiler, and emissions control devices at a power plant have to be modified to burn low-sulfur coal. The potential extent of future fuel-switching is also limited by contraints on the expansion of mining capacity and the availability of transportation facilities to move low-sulfur coals to coal consumers; however, fuel-switching in combination with other emissions control approaches can be a significant and cost-effective means of control.

A second precombustion approach is used to reduce SO_2 emissions by physically or chemically treating coal to remove some of the sulfur it contains. Sulfur in coal exists in two major forms: inorganic and organic. Inorganic or "pyritic" sulfur can be removed inexpensively through physical

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1 In-use technologies -- those being sold on a commercial scale in the U.S. that have demonstrated control capabilities.

Available technologies -- those that have been tested and proven but are not currently operational in the U.S. on any significant scale, even though they may be widely utilized in other countries.

Emerging technologies -- those still in the research and development phase, but that have been tested on at least a pilot scale.

cleaning. Organic sulfur is chemically bound to the carbon molecules of coal, and can only be removed by a chemical process. Chemical processes are in an earlier stage of development and are more expensive than physical processes, but the sulfur removal potential is higher. Chemical coal cleaning can remove greater than 90 percent of the sulfur in coal, while physical processes can only remove up to 40 percent.

II. Combustion Approaches

Currently, no combustion approach is in widespread use for the reduction of SO_2 emissions. Two emerging combustion technologies are the limestone injection multistage burner (LIMB), and fluidized bed combustion (FBC). LIMB is based on the use of staged burner techniques for NO_x control, in combination with dry limestone injection into the combustion chamber for SO_x control. EPA's LIMB research program is expected to continue through 1985, with planning goals to achieve 50 to 70 percent removal of SO_x and NO_x at a capital cost of $30-40/kW. Given continued research, it is possible that LIMB, because it may be retrofitted into existing plants at a competitive cost, may emerge as a particularly attractive control technology option.

Fluidized bed combustion (FBC) involves feeding crushed coal into a bed of inert ash mixed with limestone or dolomite. The bed is fluidized by the injection of air from the bottom of the bed. SO_2, formed during combustion, reacts with the limestone or dolomite to form a solid calcium sulfate, which can then be removed from the boiler. FBC can remove up to 90 percent of the sulfur dioxide; however, costs are very uncertain at this time.

III. Post-Combustion Approaches

Of sulfur dioxide control technologies in use today, flue gas

desulfurization achieves the highest reduction percentages for SO_2 removal. Flue-gas desulfurization (FGD) technology removes the sulfur dioxide produced during combustion by spraying the exhaust gases in the stack with a chemical absorbent, typically lime or limestone. This process is popularly referred to as "scrubbing." Of the three types of FGD systems -- wet, dry, and regenerable -- wet processes are most widely used. Wet scrubbers can efficiently remove between 70 to 95 percent of the sulfur dioxide formed during combustion. Dry scrubbing, a much newer technique, has been shown in testing to remove between 60 to 90 percent of the sulfur dioxide formed during combustion. Regenerable FGD systems are currently undergoing major research efforts. The major disadvantage of FGD systems is that they produce large amounts of sludge. A recent government report concluded that in the future the U.S. will produce more sludge from FGD scrubbing than from treating municipal sewage.

Controlling Nitrogen Oxide Emissions

Oxides of nitrogen (NOx) are formed during combustion by two processes: 1) as a result of the oxidation of nitrogen present in the fuel ("fuel NOx") and 2) by the oxidation of nitrogen in the surrounding air ("thermal NOx"). Both processes are controlled by the amount of oxygen present, while thermal NOx formation is also controlled by temperature. The proportion of thermal to fuel NOx produced during combustion varies from fuel to fuel. For coal, EPRI estimates that 20 to 40 percent of NOx emissions are "thermal" and 60 to 80 percent are "fuel."

NOx emissions can be most directly controlled by modifying combustion conditions. While the majority of NOx control techniques focus on the combustion process, post-combustion techniques (flue-gas treatment) are being

developed.

I. Combustion Approaches

For technologies in use today, the low-NOx burner achieves the highest percentages of NOx removal. Low-NOx burners, which incorporate features of the staged combustion process, have been shown to significantly reduce the formation of both fuel and thermal NOx at a relatively low cost. During the first stage of combustion, less air is supplied to the burner than is required to completely burn the fuel. Fuel-bound nitrogen is then released, but is converted to N_2 because oxidizing agents are not available. The subsequent addition of air causes the remaining fuel to be burned.

The amount by which NOx emissions can be reduced depends on the type of fuel burned, the type of boiler in use, and whether the plant is new or already exists. Emissions reductions of as much as 50 percent for retrofitting a coal-fired plant, and 60 to 80 percent for retrofitting an oil-fired plant, are achievable using the low-NOx burner technique.

II. Post-Combustion Approaches

Flue-gas treatment (FGT) is an emerging post-combustion process for high levels of NOx removal. FGT has been applied extensively in Japan for use on oil-fired boilers, but has not become as popular as the low-NOx burner in the U.S. because of its high costs for use on coal-fired boilers.

At least 50 different types of flue-gas treatment technologies are available today. Several of them are termed selective catalytic reduction (SCR). In SCR, flue gases are mixed with ammonia (NH_3) and then passed over a catalyst. The catalyst assists in the reaction of ammonia and nitrogen oxides

to form nitrogen gas and water vapor. While 90 percent NOx removal is possible using FGT, 80 percent removal is preferable in order to minimize capital and operating costs.

The following Table presents summary information on the major control technologies. Due to site-specific conditions, the removal efficiency levels given should be considered for approximation only.

TABLE 1

OVERVIEW OF CONTROL TECHNOLOGIES

Control Technology	Reduction Efficiencies	Revenue Requirements (Mills/Kwh)	Stage of Development
Sulfur Dioxide			
Fuel Switching	30-90%	0-7	In Use
Physical Coal Cleaning	5-40%	1-5	In Use
Chemical Coal Cleaning	60-85%		Emerging
Wet Flue Gas Desulfurization	70-95%	10-17	In Use
Dry Flue Gas Desulfurization	40-90%	9-15	In Use
Regenerable Flue Gas Desulfurization	70-90%	12-25	Available
Oil Desulfurization			
Indirect	30-40%		In Use
Direct	70-90%		In Use
Limestone Injection Multi-stage Burner	50 90%		Emerging
Fluidized Bed Combustion	<90%		Emerging
Nitrogen Oxides			
Low-NO_x Burner - Commercial	30-50%	0	In Use
Low-NO_x Burner - Developmental	50-80%	0	Emerging
Flue Gas Treatment Without Catalyst	35-40%		Available
Flue Gas Treatment With Catalyst	80-90%		Available

D. THE EFFECTS OF ACID RAIN CONTROL MEASURES ON THE U.S. COAL MARKET

Summary

I. INTRODUCTION

Many legislative proposals for controlling acid rain have focused on reducing sulfur dioxide emissions in the eastern United States. Sulfur dioxide is formed when sulfur, an element naturally present in fossil fuels, is "oxidized" during the combustion process. The greater the concentration of sulfur in the fuel being burned, the greater the production of sulfur dioxide gas. Consequently, switching from high- to low-sulfur coal is one way to achieve substantial emissions reductions.

By significantly increasing the demand for low-sulfur coals relative to high-sulfur coals, emissions reductions designed to control acid rain could potentially cause significant coal-market shifts. Because the sulfur content of coal varies significantly from region to region throughout the United States, any emissions reduction strategy that relies even in part on "switching" to lower-sulfur coals can potentially affect the regional distribution of coal production and employment. This analysis assesses the potential direction and magnitude of these effects due to regulations designed to control acid rain.

II. FINDINGS

A. Factors Contributing to Potential Coal-Market Shifts

The regional character of potential coal-market effects due to changes in sulfur dioxide emissions requirements stems from variations in the sulfur content of coal among various geographical areas of the United States. Figure

1 shows the distribution of the sulfur content of coal reserves for the eastern and western United States. Western reserves are primarily low-sulfur, while reserves in states east of the Mississippi are primarily high in sulfur. The exceptions to this pattern are the states of Kentucky and West Virginia, which contain significant quantities of low-sulfur reserves.

Table 1 presents each state's coal production for the utility market. The first column presents the quantity of coal, in millions of tons, produced for the utility market by each state in 1980. The second column shows the percentage of that production that would not permit utilities to comply with a 1.2 pounds of sulfur dioxide per million Btu emissions limit without applying control technologies. The third column presents the portion of this "non-compliance" coal that is sold on the spot market, i.e., that is not covered by a contract between a utility and a mining company. Non-compliance coal sold on the spot market is likely to be the most vulnerable to more stringent sulfur dioxide regulations. The final column shows the portion of non-compliance coal that is exported to other states -- an indicator of the potential efficacy of state-level policies to minimize coal market disruptions.

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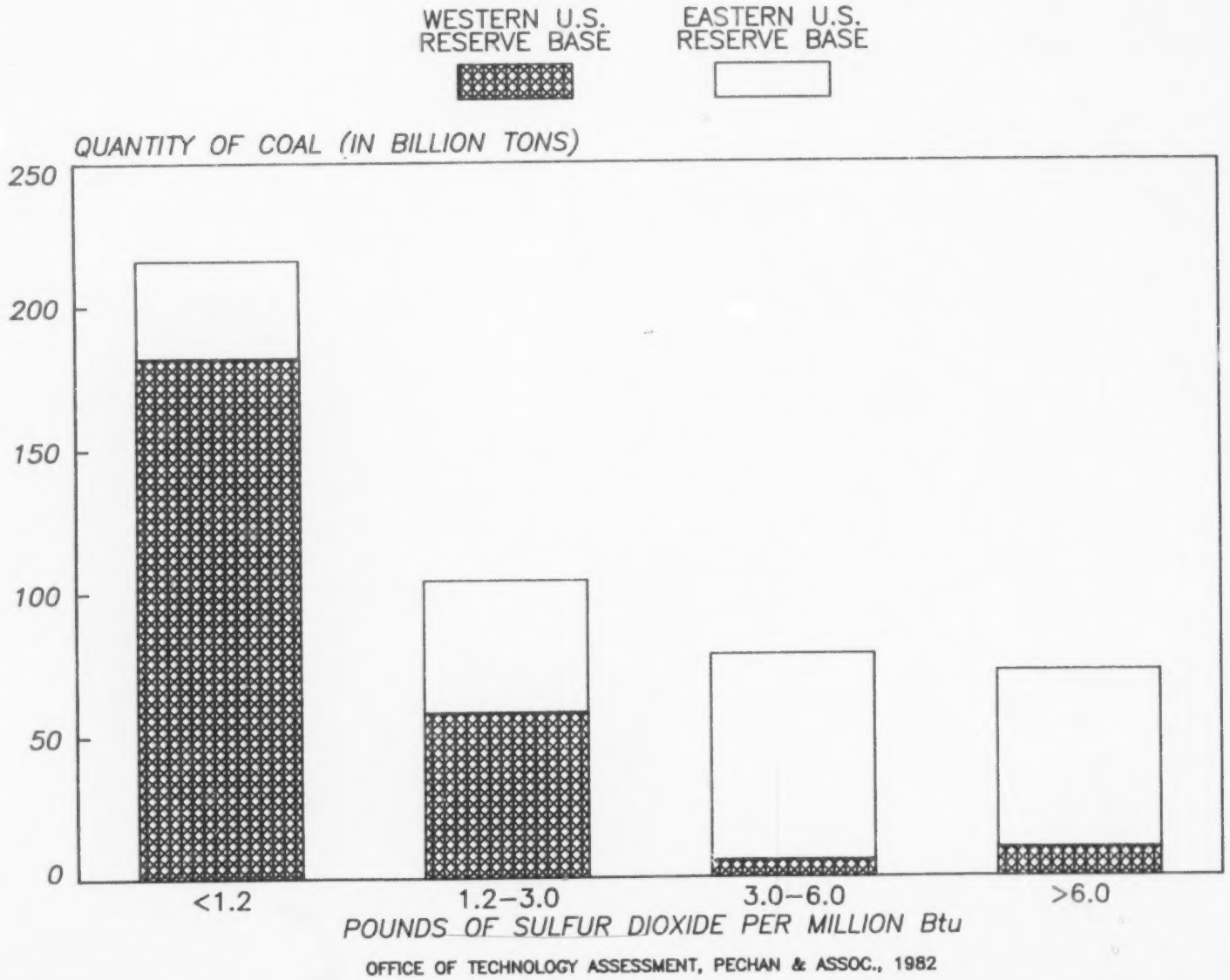
DEMONSTRATED RESERVE BASE BY SULFUR CATEGORY



OFFICE OF TECHNOLOGY ASSESSMENT, PECHAN & ASSOC., 1982

TABLE 1

COAL PRODUCTION FOR THE DOMESTIC UTILITY MARKET

	1980 Production for the Utility Market (millions of tons)	Percentage Non-Compliance*	Percentage Non-Compliance Sold on the Spot Market	Percentage Non-Compliance Exported to Other States
Alabama	15.8	82.5%	17.3%	10.1%
Arizona	10.5	0.0	0.0	0.0
Colorado	13.6	7.1	0.4	2.8
Illinois	53.4	99.9	6.6	66.0
Indiana	27.3	99.9	8.3	22.1
Iowa	0.4	95.9	14.5	0.0
Kansas	0.7	100.0	29.3	81.7
Kentucky	112.4	89.9	16.5	72.9
East	73.9	83.2	19.0	75.1
West	38.5	100.0	11.8	68.7
Maryland	1.2	100.0	22.8	24.7
Missouri	5.0	100.0	6.9	33.5
Montana	27.9	63.6	0.0	51.9
New Mexico	17.0	69.8	0.3	1.2
North Dakota	15.3	99.2	13.6	20.8
Ohio	34.3	100.0	18.9	26.5
Oklahoma	2.7	98.6	17.3	98.6
Pennsylvania	50.9	99.1	29.9	30.1
Tennessee	7.6	86.7	17.6	34.6
Texas	27.0	100.0	0.0	0.0
Utah	8.5	20.0	1.1	7.1
Virginia	13.8	87.5	10.6	71.0
West Virginia	53.1	81.3	8.1	39.4
North	30.8	97.8	7.5	46.1
South	22.3	58.5	8.8	30.1
Wyoming	89.7	16.6	0.9	9.9

* "Non-compliance" coal is defined as the amount of coal produced by a state that would not permit utilities to comply with a 1.2 pound of sulfur dioxide per million Btu emissions limit without applying control technologies.

SOURCE: From DOE/ EIA Form 423 data, supplied to OTA by E.H. Pechan and Associates.

B. Summary of Regional Analysis: Projected Patterns of Change

There is a great deal of uncertainty about how large the impacts of acid rain controls on the coal market might be. The magnitude of impact depends on the amount of emissions reductions required and the manner in which those reductions are achieved. OTA bases its estimates on results of a computer model that projects changes in regional coal production due to sulfur dioxide emissions reductions of 5, 10, and 13 million tons.* The model used, developed by ICF, Inc.,** calculates the emissions reduction strategy that minimizes costs to the utility industry. These model-based estimates are highly dependent on model assumptions of the costs of alternative emissions control approaches, projections of electricity growth, and other factors. OTA has combined ICF production projections with coal-miner productivity and income data to project the employment and economic effects of acid rain controls.

Several generalizations emerge from a review of the ICF model results. First, national coal production is projected to increase significantly by 1990. Second, regulations designed to control acid rain, even those calling for the largest emissions reductions, do not alter these projected increases at the national level. However, even though acid rain control measures are

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* In 1980 the electric utility industry in the eastern U.S. emitted 16 million tons of sulfur dioxide. Total emissions of sulfur dioxide in this region were 22.5 million tons.

** The ICF Coal and Electric Utilities Model has been modified from the DOE National Coal Model. The ICF model has been used to project the effects of acid rain control proposals by: the Environmental Protection Agency, the Department of Energy, the National Wildlife Federation and National Clean Air Coalition, and the Edison Electric Institute. The model results presented here were not performed by ICF for OTA, but for the other groups mentioned above.

not projected to produce changes in nationwide coal production, the model projects a redistribution of production among regions.

Table 2 presents the effects of acid rain control measures on regional coal production. For the high-sulfur coal areas of the Midwest (Illinois, Indiana, and Western Kentucky) and Northern Appalachia (Pennsylvania, Ohio, and Northern West Virginia), 1990 coal production is projected to be less than it would have been in 1990 without acid rain controls for all emissions reduction scenarios. For emissions reductions of 10 million tons and greater, production in these areas is projected to decline to about 10 to 20 percent below 1979 levels. Estimates of production declines are averaged over these regions, and thus may be greater or less in some states and counties than others.

For the low-sulfur coal areas of Central Appalachia (Eastern Kentucky, Southern West Virginia, Tennessee, and Virginia) and the Western U.S., acid rain control measures are projected to expand coal production beyond what 1990 levels would be, assuming no change in regulations. This effect is more pronounced in Central Appalachia than in the West, due to its proximity to eastern markets.

Employment changes in general are projected to follow changes in production. However, the uncertainty regarding future coal-miner productivity trends serves to create a larger band of uncertainty for employment effects than for production.* Table 3 displays projected effects on the coal-mine labor force of a 10 million ton sulfur dioxide emissions reduction for coal-

* Projected employment is derived by dividing coal production (tons of coal) by coal-miner productivity (tons of coal produced per miner). Therefore, for a given level of coal production, an increase in coal-miner productivity will produce a decrease in miner employment.

TABLE 2

EFFECTS OF ACID RAIN CONTROL MEASURES ON REGIONAL COAL PRODUCTION
(Percent Change in Annual Production, 1979-1990)

Region	No Change In Regulations	5 Million Ton Acid Rain Control Program	10 Million Ton Acid Rain Control Program	13 Million Ton Acid Rain Control Program
NORTHERN APPALACHIA (Maryland, Northern West Virginia, Ohio, and Pennsylvania)	+ 10%	+ 1%	- 9%	- 13%
CENTRAL APPALACHIA (East Kentucky, Southern West Virginia, Tennessee, and Virginia)	+ 49%	+ 62%	+ 70%	+ 74%
MIDWEST (Illinois, Indiana, and West Kentucky)	+ 33%	+ 12%	- 10%	- 17%
WEST	+138%	+145%	+158%	+165%

TABLE 3

EMPLOYMENT IMPACTS IN 1990 FOR THE EASTERN U.S.
OF A 10 MILLION TON SULFUR DIOXIDE EMISSION REDUCTION
(figures in parentheses are negative numbers)

	Acid Rain Case Change From 1990 Base Case[1]		Acid Rain Case Change From 1979 Level		UMW Current Unemployment[2]
	Range[3]	%	Range	%	%
Alabama	(1000-1400)	(10)%[4]	(1100)-3000	(11)-29%	6%
Illinois	(3800-11,700)	(47)	(3300-7200)	(20-44)	10
Indiana	(1200-1800)	(20)	400-2600	9-58	23
Kentucky					
eastern	5900-8600	20	(400)-15,400	(1)-43	16
western	(1200-1800)	(15)	(1200-4300)	(11-38)	
Maryland	0	0	(100)-300	(12)-27	NA[5]
Ohio	(5700-8100)	(39)	(2400-6300)	(16-42)	46
Pennsylvania	(3100-4500)	(10)	(8100)-4400	(22)-12	18
Tennessee	(1200-1800)	(22)	800-2800	24-79	0
Virginia	2300-3300	18	(1000)-5800	(6)-36	5
West Virginia					
north	(1200-1700)	(7)	(1200-8200)	(5-34)	19
south	8100-11,600	14	33,600-62,700	105-196	

1. 1990 Base Case is what 1990 employment is projected to be in the absence of changes in environmental regulations.

2. For members of the UMW only. Figures are for April 1982.

3. The ranges represent various productivity assumptions, from a 10% decrease from 1979 levels to a 30% increase.

4. Percent change from base case is independent of productivity assumptions; therefore, no range is presented.

5. Not Available.

SOURCE: Office of Technology Assessment.

producing states east of the Mississippi. Employment effects are most severe in Illinois, Ohio, northern West Virginia, and western Kentucky -- for these areas coal-mining employment is projected to decline more than 10 percent below current levels. Under a 10 million ton emissions reduction, employment in high-sulfur coal producing areas is projected to decrease by between 9,000 and 38,000 jobs over the time required to implement the reductions* (the range incorporates the uncertainty of future productivity).

These employment trends are accompanied by proportionate changes in direct miner income. At the national level, the monetary effects of coal-market shifts are minimal because benefits to low-sulfur regions are projected to balance out costs to high-sulfur regions. This level of aggregation, however, obscures the regional distribution of coal-related economic costs. Direct income effects for a 10 million ton emissions reduction are listed below:

-- Northern Appalachia: $300 to $400 million per year loss;
-- Central Appalachia: $400 to $600 million per year gain;
-- Midwest: $300 to $400 million per year loss;
-- West: $200 to $300 million per year gain.

When indirect employment and income effects -- which reflect the impact on other economic activities dependent on the coal-mining sector -- are included, total economic impacts may be 2 to 3 times greater. For the high-sulfur regions of Northern Appalachia and the Midwest, the economic costs of changes in the coal market would be added to costs of emissions reductions to electricity consumers that these same regions may incur.

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* To place these figures in perspective, there were 254,000 workers employed in the coal-mining industry nationwide in February 1982. There were 5.4 million people employed in all sectors in Illinois and 4.5 million in Ohio in April 1982.

III. ASSESSMENT OF POLICIES TO MITIGATE COAL MARKET IMPACTS

Policies are available to mitigate the effects of acid rain control measures on the coal market. Two approaches to mitigation are:

(1) The "technological approach", which requires the use of control technologies by, for example, mandating a percentage reduction of emissions regardless of the quantity of sulfur in the coal being burned; and

(2) The "local coal" approach, which involves regulating the coal consumption behavior of utilities on the basis of the location of the coal supply, as in Section 125 of the Clean Air Act.

These policies would increase utility industry costs of achieving emissions reductions, although the amount of this increase is uncertain. Besides imposing additional economic costs, these policies might also present political difficulties. In a Federal Register notice, EPA stated that Section 125 is "cumbersome, tending to create conflicts"; the agency casts "considerable doubt on the workability of this portion of the statute."

In addition to policies designed to avoid large shifts in coal markets, policies have been proposed to lessen the resultant hardship if the shifts were allowed to take place. Alternative policies might take the form of special adjustment assistance to workers, firms, and/or communities that are found to be significantly injured by the acid rain control measures.

Potential coal market shifts resulting from acid rain control measures could present difficult economic and distributional problems. Society can significantly reduce sulfur dioxide emissions and protect the high-sulfur coal industry, but only by constraining the market potential of the low-sulfur coal industry and by imposing additional costs to consumers of electricity.

Whether these increased costs are warranted by the regional economic and employment benefits that may result is a significant issue confronting the Congress in the Clean Air Act reauthorization in 1982.

E. MOBILE SOURCE EMISSIONS AND CONTROLS

Automobiles and other mobile sources produce substantial amounts of two pollutants, nitrogen oxides (NO_x) and hydrocarbons (HC). Nitrogen oxides and hydrocarbons are the primary pollutant which react in the atmosphere to form ozone and other oxidants -- secondary pollutants. NO_x can also be converted to nitrates, a component of acid deposition.

Many recent studies suggest that ozone and ozone precursors (NO_x and HC) are transported long distances. The highest concentrations of ozone do not necessarily occur where emissions densities are greatest (i.e., in cities), but, because of chemical transformations over time, at locations which are several hours downwind.

The extent to which mobile sources contribute to acid deposition is uncertain. Tailpipe emissions may not disperse sufficiently to reach the mixing layer of the atmosphere, where they are readily transported and transformed into acid compounds. However, mobile sources have been implicated in localized occurrences of acid deposition, and have been linked to increases in acid deposition in urban areas, e.g., the Los Angeles Basin.[1]

Current Emissions: Mobile sources account for major portions of both NO_x and HC emissions nationwide, and a relatively small portion of sulfur dioxide emissions. As of 1978, mobile sources contributed 40% of national NO_x emissions (10.3 million tons), approximately 38% of total HC emissions (11.8 million tons), and 3% of national SO_2 emissions (0.9 million tons).[2]

Of NO_x emissions from mobile sources, approximately 40% came from automobiles, 10% from light-duty trucks, 25% from heavy-duty trucks, and 25% from other mobile sources such as trains and off-highway vehicles. Of HC

emissions, 60% came from automobiles, somewhat greater than 10% each for light- and heavy-duty trucks, and 15% from other mobile sources.

Nitrogen oxides emissions are relatively difficult to estimate; far more direct measurement is required to develop NO_x emissions inventories than for sulfur dioxide. A comprehensive NO_x emissions estimate is currently under development by the U.S./Canadian Work Group; a preliminary inventory for highway vehicles has been developed by an OTA contractor for 1980, and is presented in Table 1.

Trends in Highway Vehicle Emissions

Since 1940, nationwide NO_x emissions have approximately tripled; a four-fold increase in the number of motor vehicles since 1945 contributed significantly to this dramatic growth rate.[3] In addition, NO_x emissions per mile driven increased during the late 1960's and early 1970's, due to the technologies used to implement the first generation of HC and carbon monoxide (CO) emissions control standards. Between 1970 and 1978, HC emissions from highway vehicles decreased by about 10%, from 11.3 to 10.2 million tons, while NO_x emissions increased by 25%, from 5.8 to 7.4 million tons.[4]

During the same period, vehicle miles travelled in the U.S. increased by about 37%, showing some decline in NO_x emission rates due to controls mandated by the Clean Air Act. Since the late 1970's, overall amounts of NO_x emissions from motor vehicles have also begun to decrease slightly as the proportion of NO_x-controlled vehicles in the U.S. increases.

To estimate future NO_x emissions levels for highway vehicles, OTA has projected three alternative travel scenarios for 1995: a no-growth, a low-growth, and a medium-growth case. Table 2 shows numbers of miles travelled

TABLE 1

1980 U.S. NO_x EMISSIONS (10^3 tons)

Eastern Region		Western Region	
State	Mobile Source	State	Mobile Source
Alabama	127	Alaska	13
Arkansas	74	Arizona	87
Connecticut	87	California	696
Delaware	20	Colorado	87
District of Columbia	13	Hawaii	20
Florida	328	Idaho	33
Georgia	194	Kansas	74
Illinois	281	Montana	27
Indiana	174	Nebraska	54
Iowa	80	Nevada	27
Kentucky	120	New Mexico	47
Louisiana	100	Oklahoma	120
Maine	33	Oregon	80
Maryland	13	South Dakota	27
Massachusetts	154	Texas	482
Michigan	281	Utah	40
Minnesota	120	Wyoming	20
Mississippi	74		
Missouri	154		
New Hampshire	27		
New Jersey	221		
New York	341		
North Carolina	187		
Ohio	321		
Pennsylvania	308		
Rhode Island	27		
South Carolina	107		
Tennessee	147		
Vermont	13		
Virginia	127		
West Virginia	54		
Wisconsin	147		
31 + D.C Total	4454	West Total	1934
		U.S. Total	6388

Source: Michael P. Walsh, "Motor Vehicle Emissions of Nitrogen Oxides", OTA contractor report, November 30, 1981.

TABLE 2

ANNUAL MILEAGE BY MODE (in millions)

Mode	1980 and 1995 No Growth***	Case 1*	Case 2**
Autos	1,131,000 (75 %)	131,091	1,762,098
Light Truck <6000	149,292 (9.9%)	173,328	232,597
Light Truck 6-8500	81,432 (5.4%)	94,543	126,871
Heavy Truck Gas	73,892 (4.9%)	54,606	54,606
Heavy Truck Diesel	57,304 (3.8%)	103,205	119,135
Motorcycle	15,080 (1.0%)	17,508	23,495

* Case 1 -- low growth

autos and light trucks grow 1% per year (1.161 x base)
heavy gasoline trucks decline 2% per year (0.739)
heavy diesel trucks grow at 4% per year (1.801)
motorcycles grow 1% per year (1.161)

** Case 2 -- medium growth

autos and light trucks grow 3% per year (1.558)
heavy gasoline trucks decline 2% per year (0.739)
heavy diesel trucks grow at 5% per year (2.079)
motorcycles grow at 3% per year (1.558)

*** Derived from data contained in MVMA Motor Vehicle Facts and Figures '81.

Source: Michael P. Walsh, "Motor Vehicle Emissions of Nitrogen Oxides", OTA contractor report, November 30, 1982.

under each of the three scenarios, and growth rate estimates for each category of highway vehicle under the two growth scenarios. Figures from the three scenarios were then used to project nationwide 1995 NO_x emissions from highway vehicles under a variety of control standards. Table 3 summarizes the effects of the various growth cases and emissions standards on 1995 NO_x emissions projections.

Two cases have been selected from among the 18 projected emissions levels to represent a likely lower and upper bound for emissions estimates. The lower-bound scenario assumes low growth, retention of the 1.0 gram per mile (gpm) automobile standard currently in effect for 1981 and later model cars, and a tightening of light- and heavy-duty truck standards to 1.2 gpm and 6.0 grams per brake horsepower-hour respectively, as the 1977 Clean Air Act Amendments require. By 1995, this would result in a 21% reduction from current NO_x mobile-source emissions.

The upper-bound scenario assumes medium growth, retention of the current light and heavy truck standards rather than any tightening, and a rollback of the automobile standard to 2.0 gpm. This would result in a 37% increase in NO_x mobile-source emissions over 1980. Assuming a low-growth scenario with these same emissions standards would result in only a 12% NO_x increase over 1980 levels.

Costs of Automobile Emissions Controls

Estimates of the costs and fuel-economy implications of emissions controls are highly controversial. The Bureau of Labor Statistics has estimated that between 1975 and 1979, pollution control devices have increased the cost of new automobiles by about $165 -- approximately 10% of the total

TABLE 3

1995 HIGHWAY VEHICLE NOx EMISSION ESTIMATES

Emissions Standards			1995 Nationwide Emissions (1000 Tons)		
Auto	Lt. Truck	Heavy Truck	No Growth	Case 1	Case 2
1.0	1.2	1.7	3067	3759	4767
1.0	1.2	4.0	3633	4608	5725
1.0	1.2	6.0	4104	5287*	6487
1.5	1.7	6.0	4764	6052	7514
2.0	2.3	10.7	5868	7469	9193**
none	none	none	7950	9880	12438

1980 Base Year Emissions = 6691 Tons

* considered most optimistic case (21% reduction from base year)

** considered most pessimistic case (37% increase from base year)

Source: Michael P. Walsh, "Motor Vehicle Emissions of Nitrogen Oxides", OTA contractor report, November 30, 1981.

cost increase for new cars during the same five years. EPA and manufacturers' estimates of the costs[5] of meeting statutory automobile emission standards of 0.4 gpm HC, 3.4 gpm CO, and 1.0 gpm NO_x are presented in Table 4. Control cost estimates vary widely; the General Motors' estimate of $720 per car is about double that of EPA, and is 50% higher than Ford's. Differences in technology among manufacturers, and the difficulty of allocating the costs of multipurpose components to particular objectives (e.g. emissions reductions, as opposed to the resulting improvements in fuel economy or vehicle performance), probably account for a substantial amount of the variation. The differences narrow when NO_x controls are looked at alone: Ford, EPA, and Chrysler show only a $27 per vehicle spread in estimates of savings associated with a change in standards from 1 to 2 grams per mile NO_x -- ranging from $48 to $75 per vehicle. GM's estimate is still substantially higher, at $188 per vehicle.

Inspection and maintenance (I/M) programs offer an alternative approach to reducing NO_x emissions from mobile sources. A recent study concluded that adding NO_x testing to an existing I/M program for HC and CO would add about $4 per inspected vehicle, and when 20% of the vehicles were repaired (at an average cost of $50 each) could result in about a 10% reduction in NO_x emissions.[6] An EPA analysis concluded that the cost-effectiveness of an I/M program for NO_x control would range from $400 to $500 per ton of NO_x eliminated if added to an existing HC/CO program, or $1700 to $2700 per ton if started up exclusively for NO_x control.[7]

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TABLE 4

COSTS OF EMISSIONS CONTROLS FOR AUTOMOBILES

Autos	Cost Differential Per Vehicle 1983 Cars vs. Uncontrolled[1]	NO_x Costs Per Vehicle 1 gpm vs. Uncontrolled[3]	NO_x Costs Per Vehicle 1 gpm vs. 2 gpm
EPA	370[2]	$123	50
Ford	500	167	48[4]
GM	725	242	188
Chrysler	---		75

1. 1983 car costs are based on emissions standards of 0.41 HC/3.4CO/1.0NO_x.
2. EPA estimates that this would drop to $330 by 1988 as systems are refined.
3. Assumes that emission control costs of 1983 cars are equally divided between HC, CO, and NO_x.
4. Assumes that 60% of cost estimated by Ford for CO, high altitude and NO_x control -- $80 per car -- is attributable to NO_x control. This is a high estimate, based on Chrysler's estimate that 60% of its costs for CO and NO_x control is due to NO_x alone.

FOOTNOTES

1. PEDCo Environmental, Inc. and Paul W. Spaite Co., Perspective on the Issue of Acid Rain, DOE Contract DE-AC21-81MC16361, June 1981.

2. National Air Pollutant Emission Estimates, 1970-1978. Office of Air Quality Planning and Standards, EPA. EPA-450/4-80-002 (January 1980).

3. Nitrogen Oxides, National Academy of Sciences, 1977.

   MVMA Motor Vehicle Facts and Figures '81, September, 1981.

4. National Air Pollutant Emissions Estimates, 1970-1978. U.S. EPA, January 1980. EPA-450/4-80-002.

5. "The Cost of Controlling Emissions of 1981 Model Year Automobiles," U.S. Environmental Protection Agency, June 1981.

   Testimony before the Health and Environment Subcommittee, U.S. House of Representatives, September 21, 1981.

   Letter from Dr. Betsy Ancker-Johnson, General Motors to Douglas Costle, U.S. Environmental Protection Agency, September 22, 1980.

6. "A study of the Relationship Between Motor Vehicle Emissions and Attainment of National Ambient Air Quality Standards," Part 1: Nitrogen Dioxide, SRI International, February 1981.

7. Internal EPA Memo. Tom Cackette to Michael P. Walsh, January 9, 1981.

F. TERRESTRIAL EFFECTS

Summary

This portion of the OTA assessment provides estimates of long-range transport air pollutant effects on agricultural and forestry resources made by researchers at Oak Ridge National Laboratory (ORNL). The section, which separately discusses ozone and acid deposition, integrates what is known about crop and forest response, the levels of the pollutants in question, and the distribution, importance, production, and value of the resource itself on a regional and national scale.

Key Findings:

The Effects of Ozone on Agriculture:

Ozone has been recognized as a factor in crop production for over two decades (Heck et al. 1977). Ozone alone or in combination with sulfur dioxide and nitrogen oxides is responsible for up to 90% of the crop losses in the United States caused by gaseous air pollutants (Heck et al. 1980). Previous estimates of crop losses were limited by lack of available field-generated data on the effects of ozone on crops, limited spatial and temporal resolution of ozone data, and an inability to integrate these factors with crop distribution and productivity data. The present analysis utilized data recently developed by the National Crop Loss Assessment Network (NCLAN) (Heck et al. 1982) in combination with more recent crop and ozone data to more accurately determine impacts on the productivity of corn, wheat, soybean, and peanut crops.

Ozone impacts on crop productivity were estimated by using new NCLAN

dose-response relationships, 1978 ozone monitoring data, and 1978 agricultural statistics. The selected crops represent a range of ozone sensitivity from sensitive (peanut) to sensitive/intermediate (soybean) to intermediate (wheat) to tolerant (corn).

The assessment estimates that in 1978, corn yields would have increased by 3%, wheat by 8%, soybean by 17%, and peanuts by 30% if ozone levels had been reduced to natural "background levels" (ambient maximum 7-hour per day concentrations below 25 ppb). As measured by 1978 crop prices, this represents about $3 billion of agricultural productivity. Of the estimated dollar impact, soybean represents 64%, corn 17%, wheat 12%, and peanuts 7%. The cornbelt states of Illinois, Iowa, and Indiana, plus Missouri, Arkansas, Minnesota, Ohio, Kentucky, North Carolina, and Virginia were estimated to have experienced the greatest agricultural effect. Figures 1 and 2 illustrate the patterns of agricultural effects of ozone as typified by projected corn and soybean yield gains. These maps show that a significant portion of the agricultural land area of the United States experiences levels of ozone during the growing season capable of reducing crop yields.

This analysis is the first of its kind and is limited by factors detailed in the assessment and by the representativeness of the data base from which it was generated. The results indicate that current ozone levels are significantly affecting agricultural productivity; continued field-scale dose-response research, however, would be required to reduce the level of uncertainty associated with this analysis.

The Effects of Acid Rain on Crops:

Acidic precipitation is frequently cited as a potential threat to

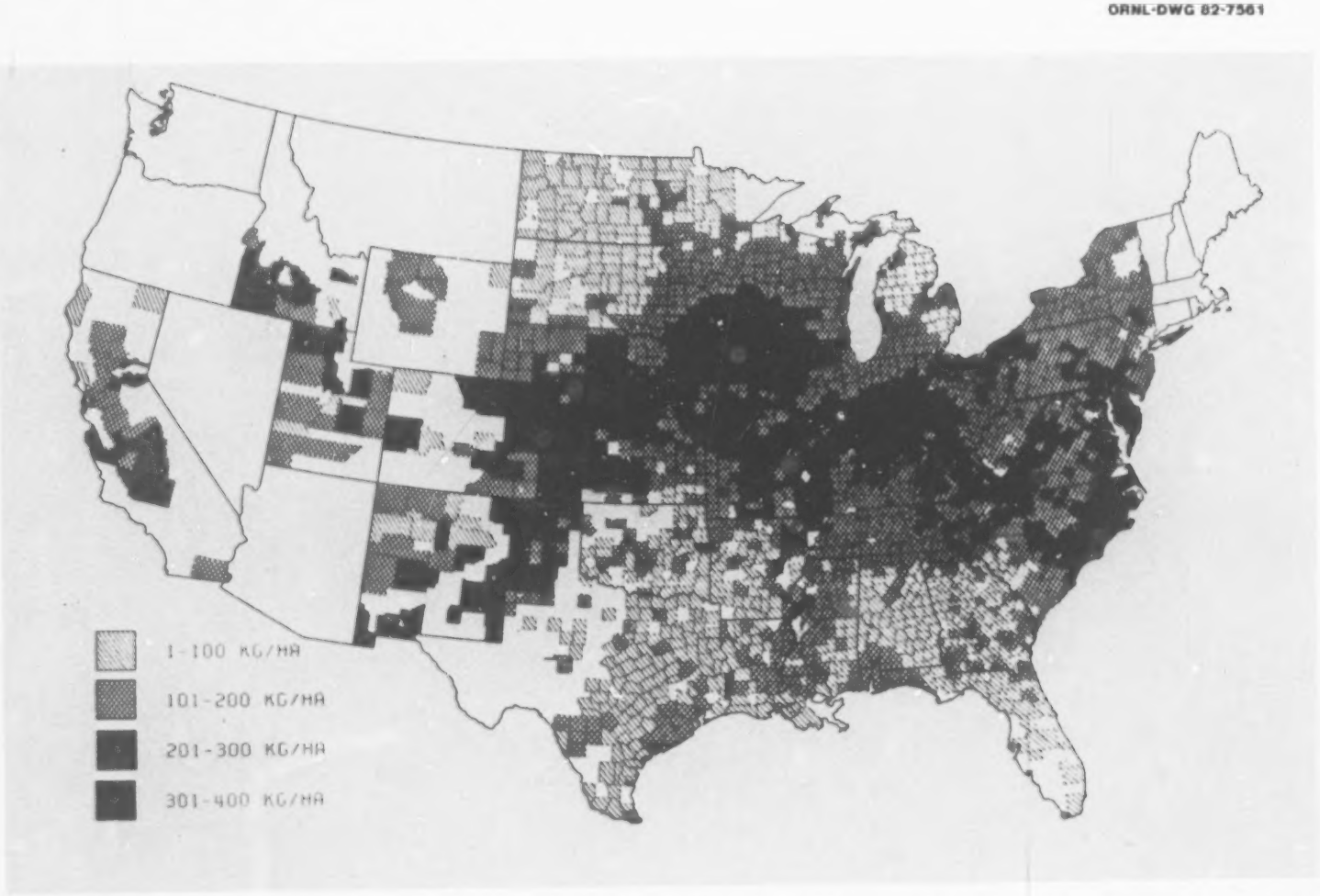

Fig. 1 . Corn yield gains (kg/ha) with ozone reductions based on 1978 agriculture census and 1978 EPA ozone data.

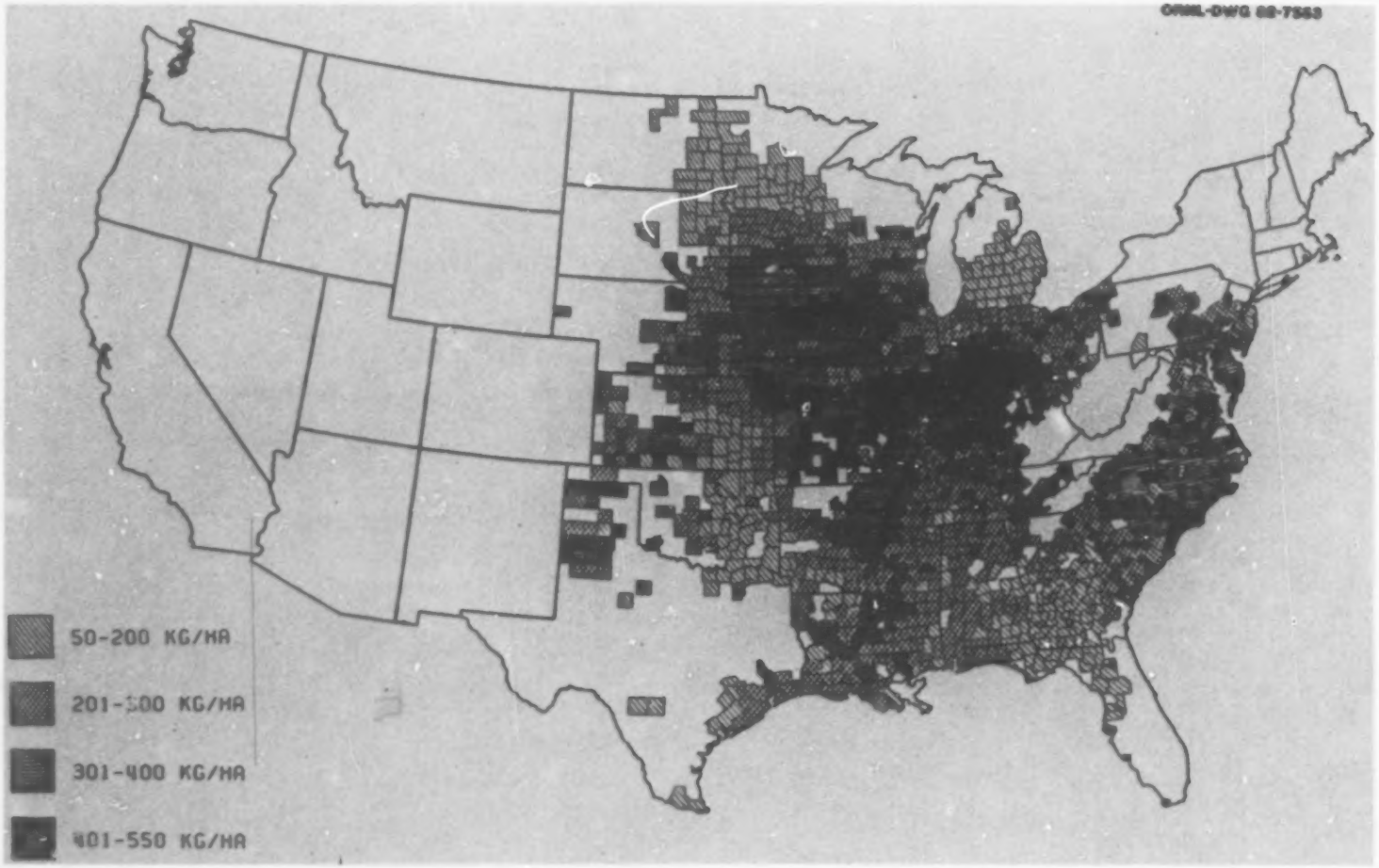

Fig. 2 . Soybean yield gains (kg/ha) with ozone reductions based on 1978 agriculture census and 1978 EPA ozone data.

agricultural productivity in the United States because patterns of deposition of "acid rain" (elevated levels of hydrogen ions) coincide with major regions of agricultural productivity. Figure 3 illustrates those agricultural regions (counties with >30% agricultural land area) that experience elevated hydrogen ion deposition. Figure 3 also shows those areas with highest ozone concentrations, to illustrate the considerable amount of regional similarity between elevated levels of both pollutants.

Crop responses to acid rain are more likely to involve direct effect on above-ground portions of the plant rather than some soil-mediated effect. Potential interactions with other physical and biotic causes of stress are hypothesized, but little definitive evidence exists on which to base conclusions.

There are no documented cases of direct, visible injury to vegetation in the field situation as a result of exposure to "acid rain." Rather, information about effects is the result of a wide variety of approaches involving, in most instances, the use of some form of rain "simulation." The evidence available clearly shows that wet deposition of pollutants may result in a range of direct effects on vegetation, depending on (1) precedent conditions (e.g., soil nutrient status, plant nutrient requirements, plant sensitivity, and growth stage) and (2) the total loading or deposition of critical ions (H^+, NO_3, SO_4).

Summarizing the current knowledge of the effects of acid deposition on crops:

(1) visible injury thresholds for acid precipitation lie between pH 2.0 and 3.6, depending on species, and may vary from pH 3.0 to 3.6 within the same

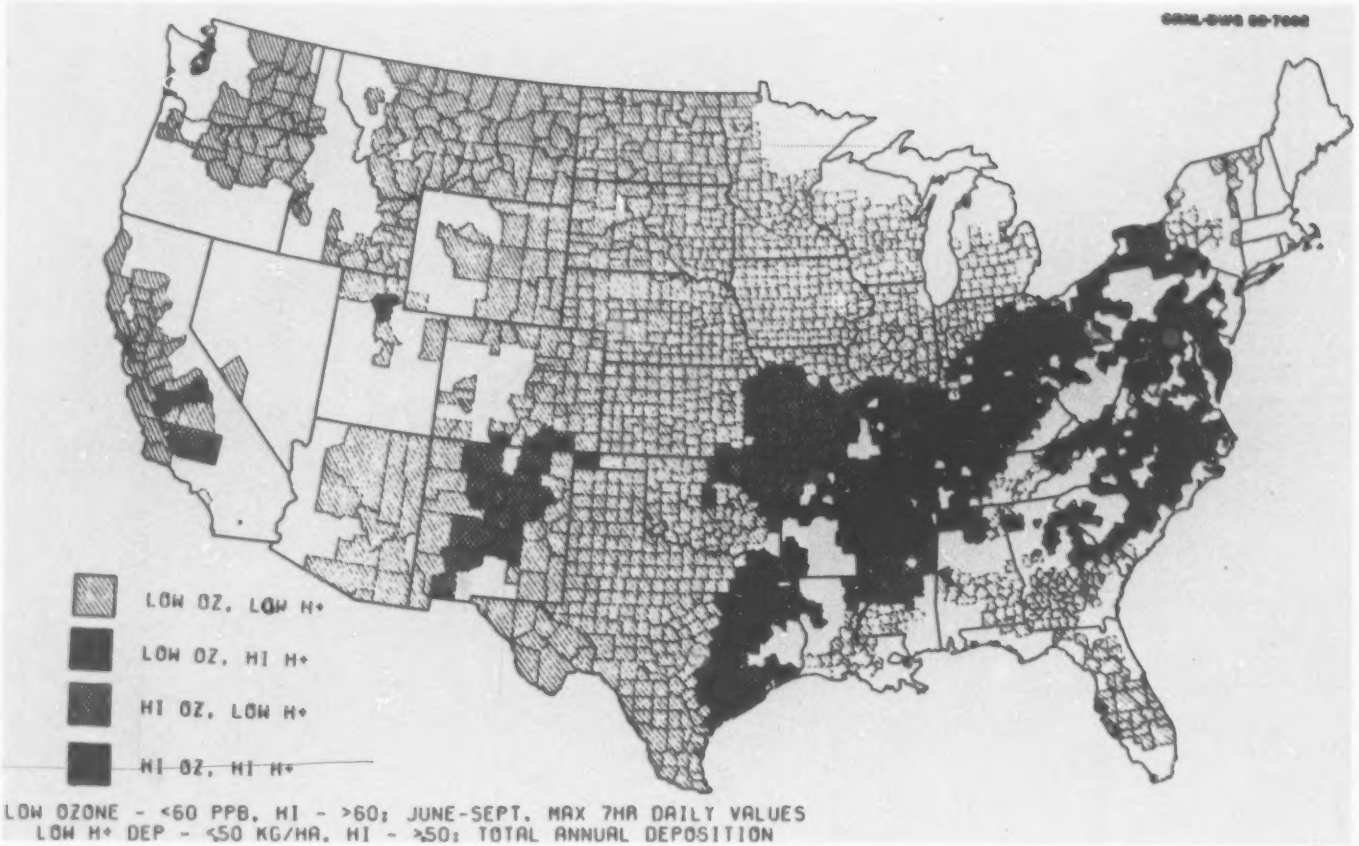

Fig. 3 Ozone and acid rain levels for agricultural areas (>30%) 1978 EPA ozone data.

species (e.g., bean);

(2) species-to-species variability in sensitivity may be as great as two pH units;

(3) the threshold for growth effects in the absence of visible injury is reported to be between pH 3.5 and 4.0, but sulfur and nitrogen inputs in the precipitation may result in a net positive growth impact, depending on soil nutrient status, buffer capacity, other growth conditions, and plant nutrient requirements; and,

(4) total dose of acidity appears to be most clearly related to visible injury.

The Effects of Acid Rain on Forests

Upon entering a forest ecosystem, acid rain becomes part of a chemical system regulated by a variety of natural processes. Hydrogen (H^+), sulfate (SO_4^{2-}), and nitrate (NO_3^-) ions are produced and cycled naturally within forest ecosystems -- acid rain should be viewed as an addition to these natural cycles. Deposition of H^+, SO_4^{2-}, and NO_3^- in excess of amounts that can be biologically utilized within the forest ecosystem will result in one of several possible situations: the soil or bedrock can (1) neutralize the acidity or trap ions so that they move no further, (2) exchange these ions for others within the soil or bedrock (for example, if the soil is acid enough, aluminum can be mobilized), or (3) allow the direct transport of sulfate and nitrate (and associated hydrogen ions) from terrestrial to aquatic ecosystems.

Recently, a significant decline in the growth of several pine species, oak, and red spruce has been noted in various parts of the northeastern United

States. In the case of red spruce, the growth decline was accompanied by considerable mortality. These trends were not accounted for by climatic patterns, and it has been suggested that acid rain may have been a factor. Though this remains a viable hypothesis in many cases, the causes for the decline in red spruce as well as in the other species mentioned above are unknown as of this writing.

Key issues in forest ecosystems exposed to acid rain are those of potential losses of certain nutrients (such as calcium and magnesium) and toxic metal mobilization. Sites most susceptible to nutrient losses are those with soils that are moderately acid (pH >5.5) and have a poor ability to store these nutrients.* Such soils occur to a very minor extent in those regions of the United States receiving significant inputs of acid rain. These soils are shown as category 1 on Figure 4 and occur extensively only in Nebraska on rangeland. Other areas of the U.S. (i.e., parts of the Adirondacks) probably have some soils of this type, but given the county level of resolution of this analysis, they are rare enough not to show up on the map. Soils moderately susceptible to acid loss are shown as categories 2 and 3 on Figure 4.

Soils most susceptible to aluminum mobilization are those with low pH (<5) -- such soils occur extensively in regions of the United States receiving significant amounts of acid rain. These soils are displayed as category 4 on Figure 4. However, these soils are frequently so naturally acidic that acid rain at current levels is unlikely to affect them significantly. The significance of aluminum mobilization to forest ecosystems is not well

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* The ability to store nutrients such as calcium and magnesium is called "cation exchange capacity". Soils considered most susceptible to nutrient losses are those with a cation exchange capacity (listed as CEC on figure 4) less than 6.2 meq/100g.

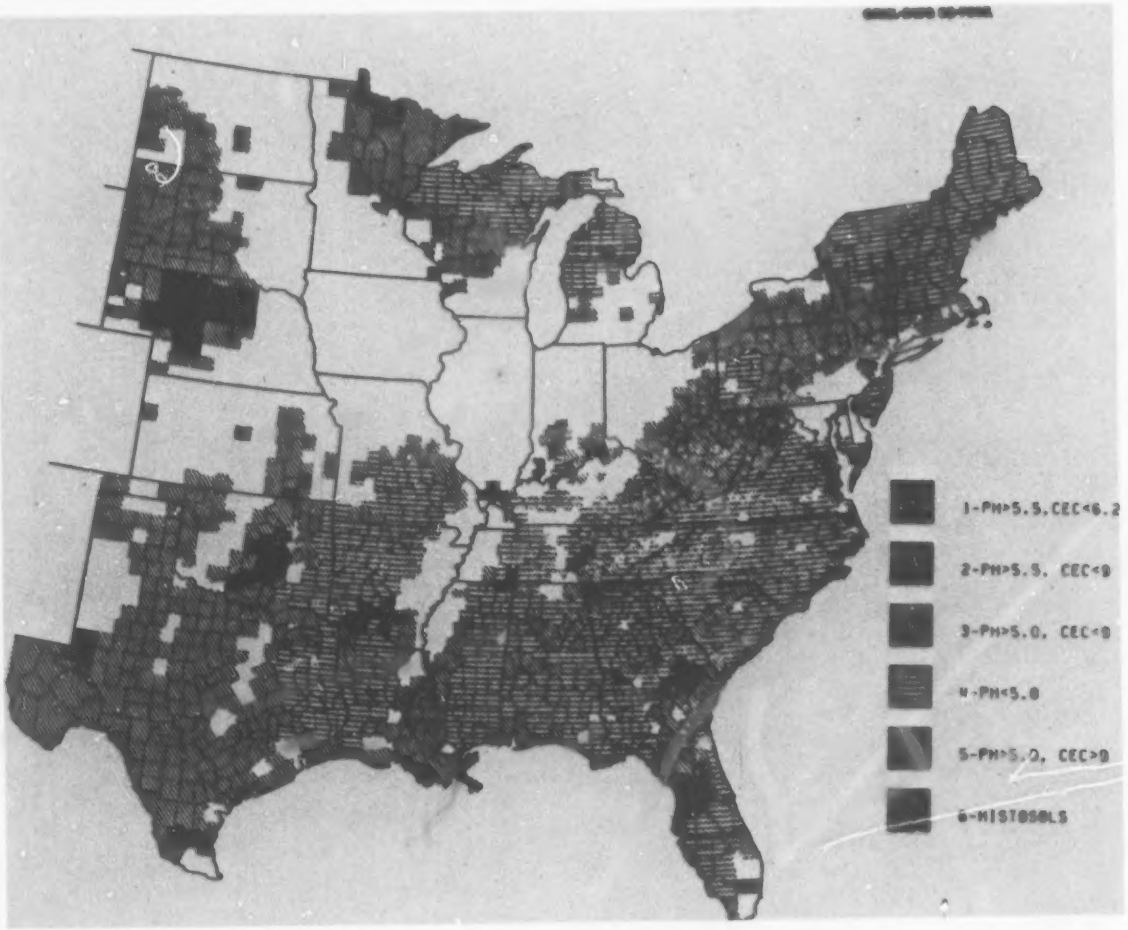

Fig. 4 . Soils characterisitics that influence the potential acidification of nonmanaged soils (<50%) 50% coverage used to classify counties soils data from 1970 National Atlas.

understood at present.

Just as acid rain may have negative effects on forests by leaching certain nutrients or mobilizing aluminum, benefits to forests may be derived by added sulfur and nitrogen. The net effect is determined by the amount of atmospheric input and the nutrient status of the affected forest sites.

Most forests of the United States are nitrogen deficient, while a few (mainly in the northeast) are deficient in nutrients such as calcium and magnesium, and a few (mainly in the northwest) are deficient in sulfur. Sulfur deposition throughout the eastern United States considerably exceeds forest sulfur requirements, whereas nitrogen deposition does not exceed forest nitrogen requirements and is probably of benefit to many forests. Temperate-climate forests typically use about 50-75 kg/ha-yr nitrogen; Figure 5 shows that deposition in these areas is about 5-10 kg/ha-yr of nitrogen.

The Effects of Ozone on Forests:

Assessing ozone effects on forest resources is difficult because of a lack of dose-response data for many important forest species, because data developed for seedlings cannot be extrapolated adequately to mature forest trees in a forest stand situation (e.g., in competition with other individuals or species), and finally because resource inventories are fragmentary and are not now available in a uniform format.

The large majority of studies of ozone effects conducted to date have dealt with the response of individual tree species rather than with the response of forests or forest ecosystems. Only one major study exists in which a substantial effort was made to examine effects to specific portions of a forest system (in the San Bernardino Mountain area of California). The

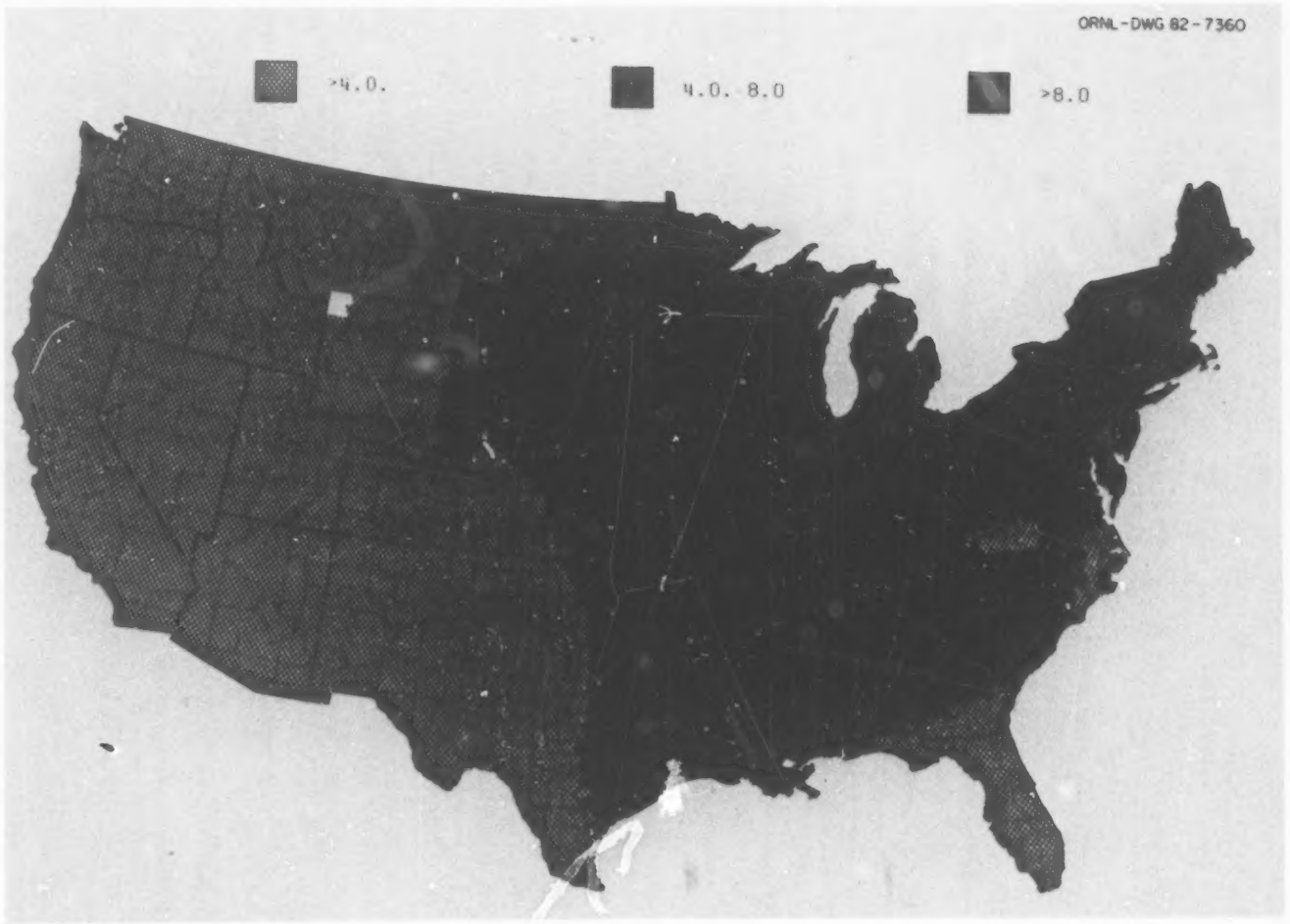

Fig. 5 . $NH_4^+ + NO_3$-N annual atmospheric deposition (kg/ha) based on the April 1979 to March 1980 period as measured by the National Atmospheric Deposition Program.

limited data available suggest that some sensitive species may experience long-term growth loss, but the quantitative evaluation of such loss is impossible at present.

A list of tree species that are sensitive, of intermediate sensitivity, and tolerant to ozone is shown in Table 1.

Interactions Between Ozone and Acid Rain -- Effects on Vegetation:

Discussion of the combined potential effects of acidic deposition and gaseous pollutants (such as ozone) on agricultural or forest productivity can only be qualitative at present. Figure 6 summarizes the various mechanisms by which agricultural or forest productivity might be affected by such a combination of pollutants. The top part of the diagram illustrates the positive and negative effects of atmospheric pollutants on exposed vegetation; the bottom part of the diagram illustrates the positive and negative effects of atmospheric pollutants on vegetation as mediated by the soil. For example, each of the gaseous pollutants ozone, sulfur dioxide, and nitrogen oxides have a predominantly negative effect on vegetation; acid deposition may have both positive and negative effects -- fertilizing the soil with sulfur and nitrogen and potentially releasing toxic aluminum within the soil.

Figure 6 indicates that the net effect on the growth of vegetation may be expected to represent the sum of a series of positive and negative effects. The net effect on plant productivity will depend strongly on those site-specific factors controlling response to any individual mechanism, assuming no interactions between mechanisms.

Negative impacts appear to reach their greatest potential in commercial forests of the southeastern coastal plains, agricultural and forest lands of

Table 1. Relative susceptibility of forest tree species to injury from ozone exposure.[a]

Sensitive	Intermediate	Tolerant
Green ash (*Fraxinus pennsylvanica*)	Boxelder (*Acer negundo*)	Black Hills spruce (*Picea glauca densata*)
White ash (*Fraxinus americana*)	Japanese larch (*Larix leptolepis*)	Colorado blue spruce (*Picea pungens*)
Quaking aspen (*Populus tremuloides*)	Black oak (*Quercus velutina*)	Norway spruce (*Picea abies*)
Honey locust (*Gleditsia triacanthos*)	Pin oak (*Quercus palustris*)	White spruce (*Picea glauca*)
European larch (*Larix decidua*)	Scalet oak (*Quercus coccinea*)	Black walnut (*Juglans nigra*)
White oak (*Quercus alba*)	Eastern white pine (*Pinus strobus*)	
Austrian pine (*Pinus nigra*)	Lodgepole pine (*Pinus contorta*)	
Jack pine (*Pinus banksiana*)	Pitch pine (*Pinus rigida*)	
Jeffrey pine (*Pinus jeffreyi*)	Scotch pine (*Pinus sylvestris*)	
Loblolly pine (*Pinus taeda*)	Shortleaf pine (*Pinus echinata*)	
Monterey pine (*Pinus radiata*)	Slash pine (*Pinus elliottii*)	
Ponderosa pine (*Pinus ponderosa*)	Sugar pine (*Pinus lambertiana*)	
Virginia pine (*Pinus virginiana*)	Torrey pine (*Pinus torreyana*)	
Hybrid poplar (*Populus maximowiezii x trichocarpa*)		
Tulip poplar (*Liriodendron tulipifera*)		
American sycamore (*Platanus occidentalis*)		

[a]Adapted from Davis and Wilhour (1976).

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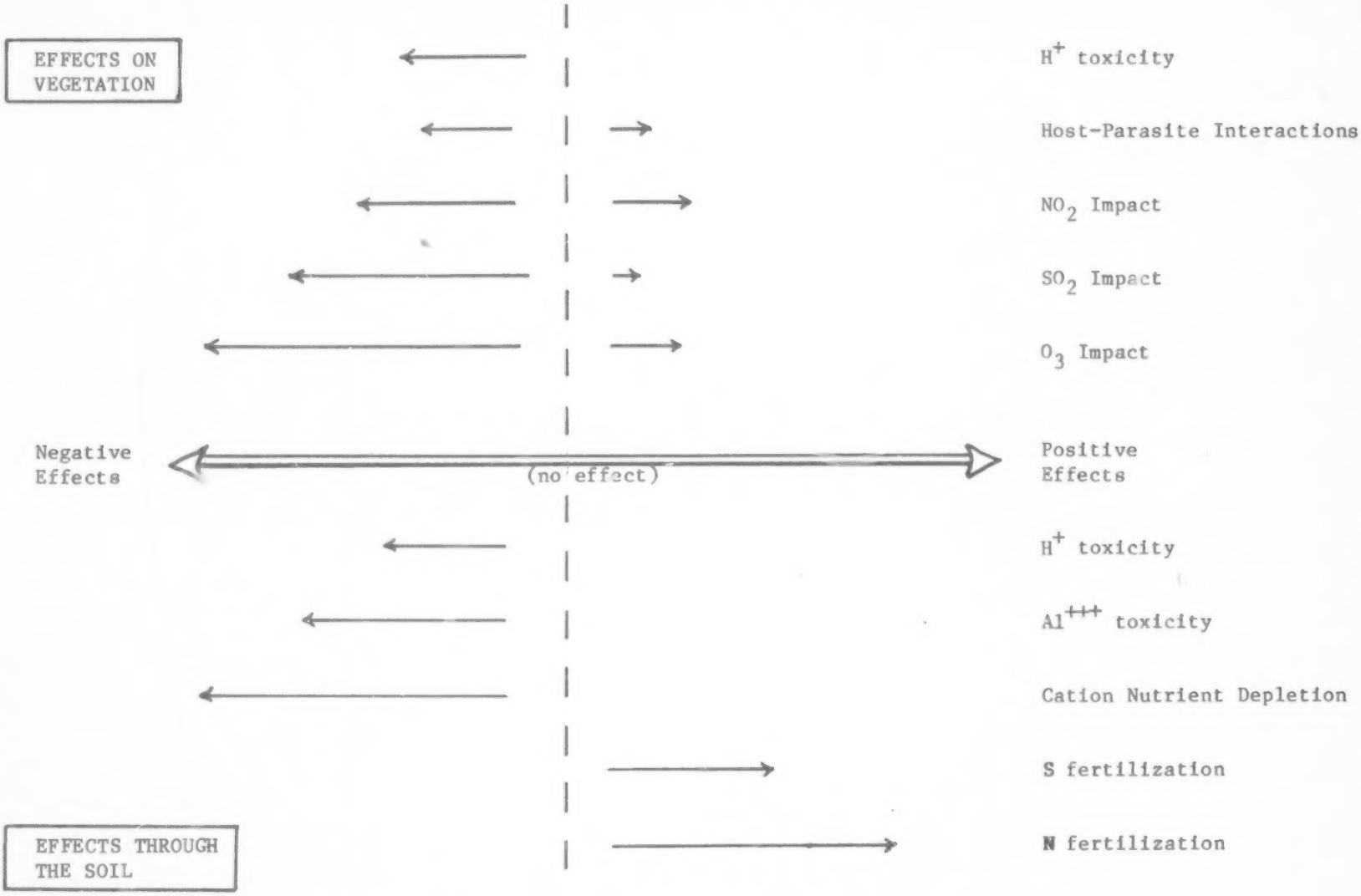

Fig. 6. A schematic representation of the mechanisms interacting to impact forest growth as a result of the combined inputs of wet and dry deposition of atmospheric pollutants.

the Mississippi River Valley, forests of the Appalachian Mountains, and agriculture and forest lands of the Ohio River Valley. In each of these regions, nitrogen inputs to forests would be expected to partially offset negative impacts, while sulfur inputs, which currently exceed forest growth requirements, probably do not represent a significant factor either positively or negatively at the present time. Such an analysis assumes that no interaction takes place between the factors influencing plant productivity. However, especially in the case of multiple gaseous pollutant exposure, some preliminary research indicates that this assumption may be invalid. At this time one can assess the potential for long-range effects on plant productivity from the combination of wet acidic deposition and dry gaseous pollutant deposition only by stating that such interactions may potentially occur and that their probability is greatest in those regions of the eastern U.S. shown as darkest areas in Figure 3 -- those with both the highest concentration of ozone and greatest deposition of acidity.

G. REGIONAL ASSESSMENT OF AQUATIC RESOURCES AT RISK FROM ACIDIC DEPOSITION

Summary

Concerns have been raised recently over the possibility that substantial numbers of U.S. and Canadian lakes and streams are being damaged as a result of acidic deposition (commonly called "acid rain"). As part of its assessment on the impact of long-range transport air pollutants, OTA contracted with The Institute of Ecology (TIE) to:

-- describe mechanisms by which acid deposition may be affecting sensitive lakes and streams;

-- provide an inventory of eastern U.S. lakes and streams considered to be sensitive to acid deposition;

-- estimate the number of lakes and streams in these sensitive regions that have been affected and/or altered by acidic deposition; and

-- examine three scenarios for future sulfate deposition levels to the year 2000, and projections of effects on sensitive aquatic resources.

Because of major scientific uncertainties and data limitations, none of these tasks can be addressed at this time with a high degree of accuracy. To illustrate, a scientific consensus has not yet been reached on the specific mechanisms by which resource damage occurs -- or on ways of quantifying the potential effects of such damage -- despite extensive research on the response of aquatic ecosystems to acid deposition. A state-of-the-art data base has been compiled, however, and plausible procedures were used to analyze that data base. Where possible, an estimate of the range of uncertainty for each of the numerical calculations has been provided.

Because of the large area of a typical watershed, most of the acid ultimately deposited in lakes and streams comes from water that runs off or percolates through the surrounding land mass, rather than from precipitation

falling directly onto water bodies. The amount of acidifying material that actually enters a given lake or stream is determined primarily by the soil and geologic conditions of the surrounding watershed. When the two major chemical components of acid rain -- nitric acid and sulfuric acid -- reach the ground, they may react with soils in a variety of ways.[1] Soils can neutralize the acids, exchange nutrients and trace metals for components of the acids, and/or hold sulfuric acid, for example.

Most soils contain amounts of counterbalancing (neutralizing) substances such as bicarbonate that may be available to "buffer" acid inputs. Such nutrients as calcium and magnesium, when present in soils, may be leached by acidic deposition and enter water bodies, while acidity remains in the soil. When soils are highly acidic, similar leaching of toxic metals such as aluminum can occur, which may cause damage to aquatic life in lakes and streams. In addition, the soils of some watersheds are able to retain sulfuric acid to varying degrees. For these areas, deposited sulfuric acid will not pass into lakes and streams until the "adsorption capacity" of the soils is exceeded.

The extent to which such reactions actually occur depends on a number of geologic and soil conditions. Where slopes are relatively steep, and soils are thin, less opportunity exists for acid precipitation to infiltrate soil layers and react chemically with soil components. Further, the composition of

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[1] For current attempts to assess the effects of acid deposition on aquatic ecosystems, sulfuric acid is considered the principal substance of concern. The importance of atmospheric inputs of nitric acid to aquatic systems is still uncertain because the nitric acid can be used as a plant nutrient. Although nitric acid deposition may influence spring acidification (depression of pH) due to snowmelt, it is unlikely to have an appreciable effect on midsummer acidity.

soils -- the availability of neutralizing material, exchangeable elements, or sulfuric acid adsorption capacity -- also affects their overall ability to mitigate acid deposition inputs to water bodies. Watersheds in a number of regions are believed to provide aquatic resources with virtually complete protection from current levels of acid deposition. In other regions, however, watersheds have little capacity to neutralize acidic substances, and much of the acid in precipitation in these regions moves through the watershed into lakes and streams.

Under unaltered conditions, virtually all lakes and streams also have some acid-neutralizing capacities. Like soils, their waters contain such substances as bicarbonate, which neutralize the entering acids. Alkalinity levels, which are expressed in microequivalents per liter of water (ueq/l), measure the acid-neutralizing capacity of lakes and streams. Lakes and streams defined as very sensitive to acidic inputs have alkalinity levels of 0-40 ueq/l, while a lake with high capacity for acid neutralization can measure over 500 ueq/l.[2] Alkalinity levels in lakes and streams also depend on geochemical conditions in the surrounding watershed and underlying bedrock. If the surrounding watershed contains little neutralizing material,

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2 Reports by Kramer (1976), Harvey et al. (1981) and others have divided the spectrum of their sensitivity measure into a series of intervals referred to as sensitivity classes or categories. Each approach differed slightly from the others, leading the authors of the U.S.-Canada Aquatic Impacts Assessment Subgroup (U.S.-Canada 1981) to propose the following classification for lakes:

Class		Alkalinity (ueq/l)
I	Acidified	≤ 0
II	Extreme Sensitivity	0 - 39
III	Moderate Sensitivity	40 - 199
IV	Low Sensitivity	200 - 499
V	Not Sensitive	≥ 500

natural alkalinity levels in lakes and streams may be quite low, making these aquatic resources highly sensitive to even low levels of acidic inputs.

When acids enter a lake or stream, available neutralizing substances are consumed, and alkalinity levels are depressed. A lake or stream is considered to be acidified when no neutralizing capacity is left. Such bodies of water have been measured as having negative alkalinity levels as great as -100 ueq/l. When water bodies become acidified, aquatic plant and animal populations may be altered.

To develop an inventory of sensitive (but not necessarily altered) lakes and streams in the eastern U.S., a multi-step process was employed. TIE (adapting a procedure developed by Oak Ridge National Laboratory) first used data on soil, watershed, and bedrock characteristics in 27 eastern states to generate a list of individual counties whose freshwater resources were likely to be sensitive to acidic deposition. The list of counties was then aggregated into 14 acid-sensitive regions (Figure 1). Using U.S. Geological Survey maps, the contractor systematically sampled each region to estimate its total number of lakes greater than 15 acres in size, and total miles of first- and second-order streams.[3] The results show an estimated 17,000 lakes and 117,000 miles of first- and second-order streams in the fourteen sensitive regions (Table 1).

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[3] The smallest unbranched tributary of a stream is a first-order stream; the junction of two first-order streams produces a stream segment of order 2.

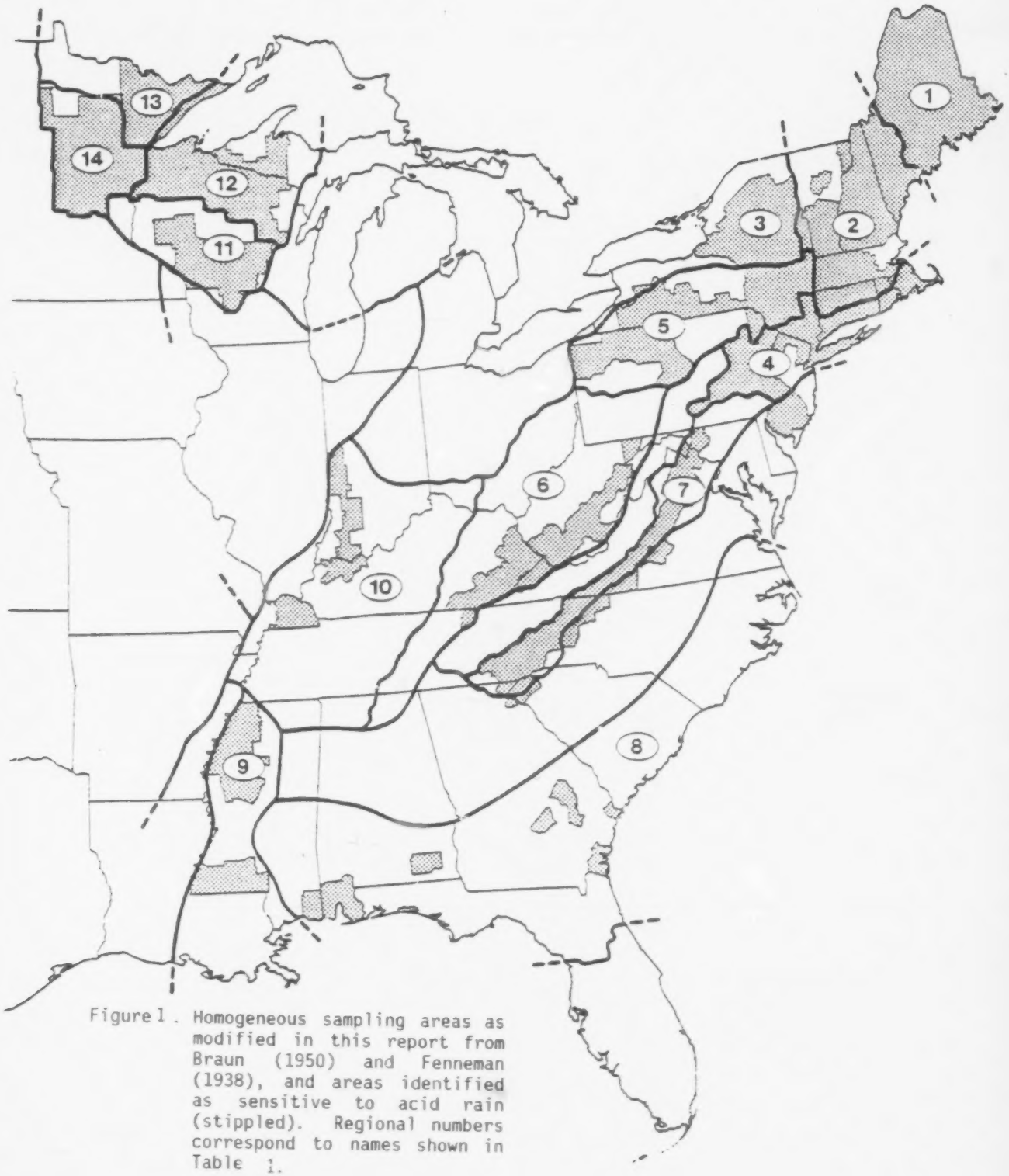

Figure 1. Homogeneous sampling areas as modified in this report from Braun (1950) and Fenneman (1938), and areas identified as sensitive to acid rain (stippled). Regional numbers correspond to names shown in Table 1.

Table 1. Total lake and stream resources estimated for the acid sensitive regions of the Eastern U.S. (see Figure 1.) Lake estimates have a reproducibility of 10 to 20%, and stream estimates approximately 10%.

Region	Sensitive Area (mi^2)	-Total Lakes- Number	Acres	-Total Streams (mi)- 1st Order	2nd Order
1. Eastern Maine	26,398	1,425	582,825	9,714	3,485
2. Western New England	29,666	1,543	763,785	15,308	4,569
3. Adirondacks	14,066	1,139	231,217	5,289	3,024
4. East Pennsylvania S. New England	20,947	1,320	118,800	8,400	2,556
5. W.New York/Pennsylvania	25,051	376	16,920	7,114	1,678
6. Appalachian Plateau	16,190	13	29,510	7,350	2,299
7. Blue Ridge/ Great Smoky Mts.	20,964	126	14,868	10,901	3,396
8. Coastal Plain	9,264	241	8,917	1,547	713
9. Lower Mississippi	13,075	170	56,610	5,374	1,255
10. Indiana/Kentucky	8,603	9	603	2,805	989
11. Central Wisconsin	12,141	583	187,726	2,683	728
12. Wisconsin/Michigan Highlands	19,229	5,307	801,357	5,037	1,737
13. Northeast Minnesota	10,560	1,637	473,093	1,637	475
14. Central Minnesota	18,870	3,170	323,340	4,831	2,529
TOTALS	245,024	17,059	3,609,571	87,990	29,433

Of this total number of lakes and streams in the sensitive regions, only a portion may be considered sensitive to acid inputs. Local variations in geology, soil conditions, and runoff patterns cause alkalinity levels of lakes and streams within a sensitive region to range from less than zero ueq/l (acidified), to over 500 ueq/l (acid resistant). Regional water quality surveys were used to estimate the percentage of the lakes and streams in the 14 regions that can be considered sensitive to acid deposition, using a 200 ueq/l alkalinity level as the cut-off point between sensitive and non-sensitive lakes and streams. Results show an estimated 9,400 lakes and 60,000 miles of streams that are either currently sensitive to or have previously been altered by acid inputs. (Tables 2 and 3).

To estimate the portion of these sensitive lakes and streams that have already been altered by acidic deposition, TIE compared data on the distribution of alkalinity levels for two distinct geographic areas: a) regions with little neutralizing capacity that currently receive high (resource-affecting) levels of acidic deposition, and b) geologically similar areas of northwestern Ontario and northern Minnesota having little neutralizing capacity, but that receive negligible amounts of acid deposition. The difference between the two distributions provides an estimate of the proportion of lakes and streams that can be considered "acid-altered," rather than simply sensitive. Only lakes with alkalinity levels less than 40 ueq/l and streams less than 100 ueq/l alkalinity -- already acidified or extremely sensitive to further alteration -- are considered in these calculations. Calculations using this approach show an estimated 3000 lakes and 23,000 miles of streams with alkalinity levels that can be described as already acid-altered. This corresponds to 18% of the lakes and 20% of the streams located within the 14 sensitive areas. See Tables 4 and 5.

Table 2. Total lake resources present and at risk in the acid sensitive regions of the Eastern United States.

Region	Sensitive Area (mi^2)	Total Number Lakes (±10%)	Percentage of Lakes <200 ueq/ℓ (± 10-20%)	Calculated No. Lakes at Risk (±20%)
1. Eastern Maine	26,398	1,425	(80)[1]	1,140
2. Western New England	29,666	1,543	(80)[1]	1,234
3. Adirondacks	14,066	1,139	80	911
4. East Pennsylvania S. New England	20,947	1,320	(80)[1]	1,056
5. W.New York/Pennsylvania	25,051	376	(80)[1]	301
6. Appalachian Plateau	16,190	13	--[4]	--
7. Blue Ridge/ Great Smoky Mts.	20,964	126	--	--
8. Coastal Plain	9,264	241	--	--
9. Lower Mississippi	13,075	170	--	--
10. Indiana/Kentucky	8,603	9	--	--
11. Central Wisconsin	12,141	583	(42)[2]	245
12. Wisconsin/Michigan Highlands	19,229	5,307	42	2,228
13. Northeast Minnesota	10,560	1,637	48	786
14. Central Minnesota	18,870	3,170	(48)[3]	1,522
TOTALS	245,024	17,059		9,423

1. In the absence of other data, alkalinities from the Adirondacks have been used.

2. In the absence of other data, alkalinities from N. Wisconsin have been used.

3. In the absence of other data, alkalinities from N. Minnesota have been used.

4. No estimate is being made for regions fewer than 250 lakes.

Table 3. Total first- and second-order stream resources present and at risk in the acid-sensitive regions of the Eastern United States. The confidence interval which would include 95% of replicated estimates is estimated to be ± 20%.

Region	Sensitive Area (mi^2)	Total Streams (mi) 1st Order	2nd Order	Percentage of Stream (± 10%) <200 ueq/l	Calculated Total Miles (± 20%) of 1° Streams at Risk	2° Streams at Risk
1. Eastern Maine	26,398	9,714	3,485	81	7,868	2,823
2. Western New Eng.	29,666	15,308	4,569	56	8,573	2,559
3. Adirondacks	14,066	5,289	3,024	(56)[1]	2,962	1,693
4. East Penna. S. New Eng.	20,947	8,400	2,556	38	3,192	971
5. W. New York/Penna.	25,051	7,114	1,678	61	4,340	1,024
6. Appalachian Plat.	16,190	7,350	2,299	(61)[2]	4,484	1,402
7. Blue Ridge/ Smoky Mts.	20,964	10,901	3,396	64	6,977	2,173
8. Coastal Plain	9,264	1,547	713	(43)[3]	665	307
9. Lower Miss.	13,075	5,374	1,255	(43)[3]	2,311	540
10. Indiana/Kentucky	8,603	2,805	989	(43)[3]	1,206	425
11. Central Wis.	12,141	2,683	728	43	1,154	313
12. Wisconsin/Mich. Highlands	19,229	5,037	1,737	13	655	226
13. Northeast Minn.	10,560	1,637	475	(13)[4]	213	62
14. Central Minn.	18,870	4,831	2,529	(13)[4]	628	329
TOTALS	245,024	87,990	29,433		45,228	14,847

1. In the absence of other data, alkalinities from W. New England and W. New York/Pennsylvania have been used.

2. In the absence of other data, alkalinities from W. New York/Pennsylvania have been used.

3. In the absence of other data, alkalinities from Central Wisconsin have been used.

4. In the absence of other data, alkalinities from Northern Wisconsin have been used.

Table 4. An estimation of the "acid-altered" lake resources of the eastern United States (only lakes with alkalinity less than 40 ueq/l) using 1980 calculated alkalinity depression and "historic" area alkalinity distributions as a control.

Region	Total No. Lakes	(1980) No. Lakes (Alkalinity <40 ueq/l)	Historic No. Lakes (Alkalinity <40 ueq/l)	(1980) No. Lakes Altered (Alkalinity <40 ueq/l)
1	1425	456	43	413
2	1543	494	46	448
3	1139	456	34	422
4	1320	423	40	383
5	376	150	11	139
6	13	----	----	----
7	126	----	----	----
8	241	----	----	----
9	170	----	----	----
10	9	----	----	----
11	583	134	17	117
12	5307	1220	159	1061
13	1637	49	49	0
14	3170	95	95	0
TOTAL	17059	3477	494	2983

Total number of lakes altered, Class I and II: 2,983

Percentage of lakes altered: 18%

Table 5. An estimation of the "acid-altered" stream resources of the eastern United States (only streams with alkalinity less than 100 ueq/l) using both calculated alkalinity depression and alkalinity distribution from untreated areas as a baseline or control.

Region	Total Stream Miles	1980 Miles (Alkalinity <100 ueq/l	Historic Stream Miles Alkalinity <100 ueq/l	Stream Miles Altered Alkalinity <100 ueq/l
1	13,199	3,746	816	2,930
2	19,877	5,641	1,228	4,413
3	8,313	3,060	514	2,546
4	10,956	3,109	677	2,432
5	8,792	3,237	543	2,694
6	9,649	2,738	596	2,142
7	14,297	4,058	884	3,174
8	2,260	482	140	342
9	6,629	1,412	410	1,002
10	3,794	809	234	575
11	3,411	726	211	515
12	6,774	418	418	0
13	2,112	130	131	0
14	7,360	455	455	0
TOTAL	117,423	30,021	7,257	22,765

1. To correct for alkalinity distributions between streams and lakes, all streams less than 100 ueq/l were included in the calculations.

2. Total stream miles altered, Class I, II and 38% of III: 22,765, representing 20% of the total first and second order stream resource.

Finally, TIE employed a simple model to estimate the effects on aquatic resources of three possible future acid deposition scenarios. As an underlying assumption, the model uses an empirical measure to project changes in lake and stream alkalinities given a change in sulfate deposition. This measure -- the "alkalinity impact parameter" -- is based on both limited observations through time of alkalinity changes in areas where sulfate deposition has been increased or reduced, and on current theory about the processes involved.[4] Although not yet fully tested, the model suggests that lake and stream alkalinities are likely to respond to future changes in sulfate deposition.

For scenario #1, a 10% increase in sulfate deposition by the year 2000, the model projects that 5 to 15 percent (depending on the region) of the most sensitive lakes and streams worsen in condition -- becoming either "acidified" or "extremely sensitive" to acid inputs. Under scenario #2, which provides

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[4] An empirical "alkalinity impact parameter" (AIP) can be defined as the ratio of observed changes in lake and stream alkalinity levels to sulfate loadings to the watershed. When sulfate retention by the watershed approaches zero, nitrate uptake by plants is high, and further losses of nutrient cations -- for example, calcium and magnesium -- are low or negligible, acidity (H_2SO_4) directly affects alkalinity (HCO_3^-), and the expected AIP is 2. For the case where some neutralization of acid input occurs in the watershed, the AIP would be less than 2; during the very early stages of acidic deposition it should approach zero. Where soils have high sulfate adsorption capacity, the AIP would also approach zero during the early stages of an acidification process; later, if acid deposition to the watershed is decreased, some of the sulfate previously retained can be washed from the soil and AIP values could exceed 2. Calculations using an AIP of greater than 2 were not considered, as many of the watersheds in the regions mapped as sensitive for this study have low sulfate adsorption.

The importance of atmospheric deposition of calcium and magnesium has been considered. For the areas examined, calcium and magnesium deposition appears to be only one-tenth the magnitude of sulfate deposition, therefore, their role has been considered negligible for the present calculations.

for a 20% decrease in sulfate deposition by the year 2000, 10-25% of the most sensitive lakes and streams are estimated to experience some recovery -- i.e., "acidified" aquatic resources become "extremely sensitive" or "extremely sensitive" resources become "moderately sensitive". Scenario #3, which involved a 35% decrease in sulfate deposition by the year 2000, is estimated to result in some recovery for 14 to 40% of the most sensitive aquatic resources.[5]

A somewhat different approach to summarizing the projected effects of Scenarios 1, 2, and 3 on surface water resources is presented in Figures 2 and 3. Scenarios 1, 2, and 3 are shown in that order in each bar-graph. In these two figures the change being shown is the sum of the calculated shifts to or from the most sensitive resource categories, expressed as a percent of the total lake or stream resource in each area. The short, solid portion of each bar represents a highly probable response; the shaded portion of each bar represents the upper bound of the probable response.

The responses under Scenario 3 by the year 2000 (a 35% reduction in SO_4^{2-} loadings) appear likely to be relatively significant for the total resource. Recovery is projected to range from a few to about 30% of the lakes in each area and up to 20% of the streams, with the greatest recovery in areas of highest current deposition.

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5 Because the TIE model is most applicable to those lakes and streams in the acid-altered (Class I: < 0 ueq/l) and extremely sensitive (Class II: 0-40 ueq/l) categories, transfers into and out of each of these categories was adopted by TIE as the measure of change of the aquatic resource in response to altered deposition. To correct for differences in alkalinity distributions between streams and lakes, a defined portion of the streams in Class III (40-100 ueq/l) was included in the TIE calculations.

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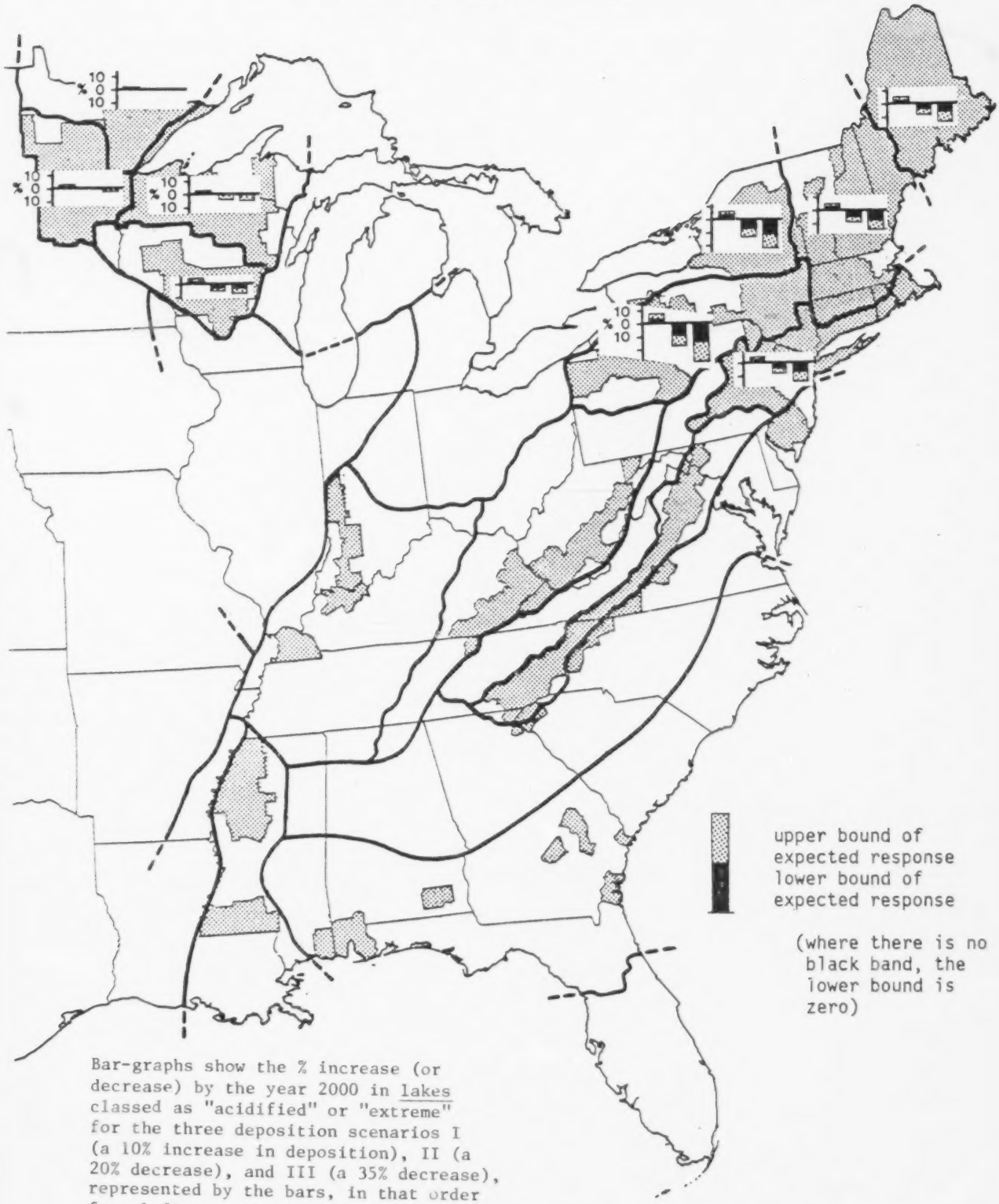
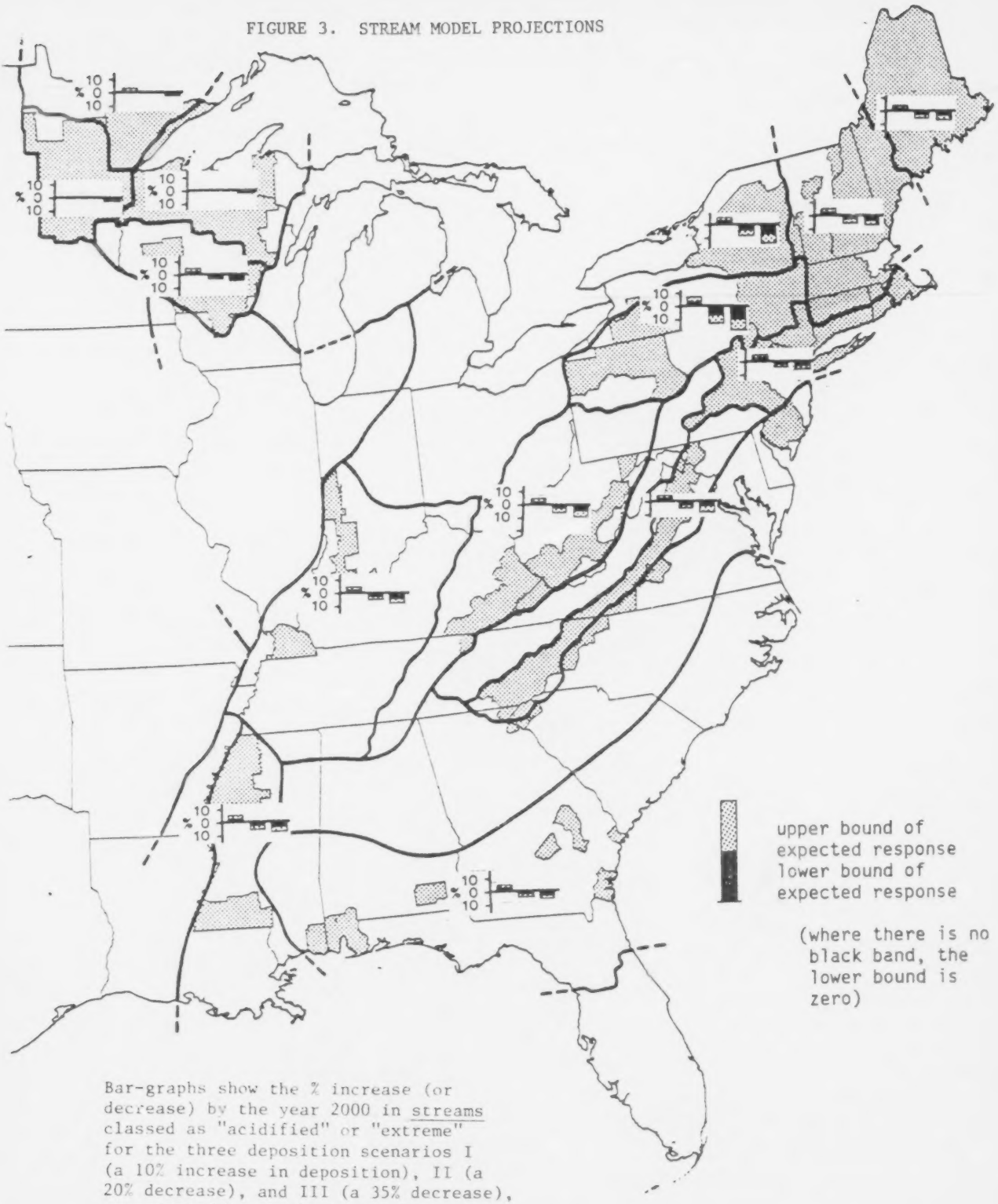
FIGURE 2. LAKE MODEL PROJECTIONS


10
0
10
upper bound of expected response
lower bound of expected response
(where there is no black band, the lower bound is zero)

Bar-graphs show the % increase (or decrease) by the year 2000 in lakes classed as "acidified" or "extreme" for the three deposition scenarios I (a 10% increase in deposition), II (a 20% decrease), and III (a 35% decrease), represented by the bars, in that order from left to right.

FIGURE 3. STREAM MODEL PROJECTIONS


10
% 0
10
upper bound of expected response
lower bound of expected response
(where there is no black band, the lower bound is zero)

Bar-graphs show the % increase (or decrease) by the year 2000 in streams classed as "acidified" or "extreme" for the three deposition scenarios I (a 10% increase in deposition), II (a 20% decrease), and III (a 35% decrease), represented by the bars, in that order from left to right.

While the model indicates some important prospects for recovery, it cannot address certain long-term consequences of acidification. Soils which may have been depleted of nutrients, such as calcium and magnesium, by acid deposition may take a great deal longer to recover normal nutrient levels than water bodies take to regain equilibrium alkalinity levels. If the soil's ability to mitigate acid deposition recovers slowly, "acid shock" episodes to water bodies from such events as spring snowmelt may persist for some time. The magnitude of such consequences is unpredictable at this time.

Lastly, some discussion of the implications of these results for biological responses in aquatic systems is required. Scientists are gradually developing an understanding of how fish and other aquatic life are affected by acid-induced alterations of their surrounding environments. Thus, predictive statements about changes in water quality measures over some period of abatement represent the first step in making predictive statements about the potential recovery of aquatic life.

H. ALTERATIONS TO AQUATIC ORGANISMS ASSOCIATED WITH ACID DEPOSITION

Losses of fish populations attributed to the effects of acid deposition have received a great deal of public attention; however, the available evidence suggests that acidic waters also affect many other forms of aquatic life, from single-celled algae, to large aquatic plants, to amphibians such as frogs and salamanders. Adverse effects on aquatic plants and animals can affect the availability of food to other animals such as fish, aquatic birds, and mammals.

Fish reproduction requires water pH levels of above 4.5, according to numerous laboratory studies and field surveys.[1] The International Joint Commission has recommended a water quality standard of greater than 6.5 pH for successful fish reproduction. Death of adult fish does not generally occur until the pH is less than 5.0. Rapid decreases in stream and lake pH due to spring snowmelt and release of acid accumulated over the winter can be detrimental if they coincide with sensitive periods of the fish reproductive cycle. Several reports have documented sudden fish kills in both rivers and lakes associated with springtime pH depressions.[2]

Survival of fish in water of low pH is influenced by temperature, presence of metals such as aluminum, hardness of the water, and type of acid input. Aluminum has been found to increase the sensitivity of fish to low pH levels.[3] Increases in the concentration of aluminum are correlated with decreasing pH. Fish mortalities have been documented as a result of increased aluminum concentrations, increased acidity, and the combination of these two factors. The hardness of the water (mineral content) increases the ability of fish to withstand low pH; fish communities disappear from soft waters at higher pH than they do from hard waters.[4]

A recent inventory (1980)[5] of the New York State Adirondacks (one of the largest sensitive lake districts in the eastern U.S. receiving significant amounts of acid deposition) indicates that the brook trout fishery has been most severely affected by acidification. At least 180 former brook trout ponds in the Adirondacks will no longer support populations. A survey of 214 Adirondacks lakes in 1975 revealed that 52% had surface pH levels below 5.0, and that 90% of these were entirely devoid of fish life. Some of these lakes had been surveyed between 1929 and 1937, when only 4% (or 10 lakes) were below pH 5.0 and devoid of fish; over the intervening forty years entire fish communities of brook trout, lake trout, white sucker, prawn bullhead, and several cyprinid species were eliminated.[6]

Similar losses of fish species have been observed in acidic lakes in the La Cloche mountain range of Ontario, Canada.[7] These field studies performed over time show that species vary in their susceptibility to declining pH, and that the mechanisms by which individual species are eliminated are complex. In Nova Scotia there are 9 rivers with pH less than 4.7 which previously had salmon that can no longer sustain trout or salmon reproduction.[8] Losses to brook trout populations in the Great Smokey Mountain National Park have also been associated with the acidity of streams and aluminum concentrations.[9]

In the field, mass mortalities of fish have been observed during episodes of spring acidification due to snowmelt. Elevated concentrations of aluminum mobilized from the soils by strong acids present in snow meltwater are thought to be a contributing factor to such large-scale fish mortality.[10] Increases in juvenile salmon mortality in Nova Scotia hatcheries have also been

associated with snowmelt-induced pH depressions.[11]

Field surveys and laboratory experiments have shown amphibian populations, such as frogs and salamanders, to be extremely sensitive to changes in pH. Many species breed in temporary pools that may be formed from low-pH meltwater in spring. Because of the great vulnerability of their habitat to pH depressions, damage to amphibian populations may be one of the earliest consequences of acidification of freshwaters. Experiments have shown correlations between pH and both mortality and embryo deformity in frog and salamander populations.[12]

Numerous macroinvertebrate animals are known to be affected by the acidification of water, although individual species may vary greatly in sensitivity. Of these, shell-bearing organisms and molting crustaceans appear to be the most sensitive to low pH.[13] No molluscs are known to inhabit waters of pH lower than 6.0, while most crustaceans (e.g., crayfish) are absent from waters of pH below 4.6.[14] Aquatic insects exhibit a wide range of sensitivities to pH.[15]

Single-celled algae are a basic constituent of the aquatic food chain. Studies have shown that as pH decreases, significant changes occur in the species and diversity of algae that predominate.[16] As lakes or streams acidify, acid-tolerant algae proliferate.[17] This group of algae is not readily edible by zooplankton, the animals that link algae and smaller fish in the food chain. However, a recent study suggests that algal species have some capability to adapt to acidic environments over the long term. This may explain the observation that a group of relatively recently-acidified lakes in Norway have less diverse algae than natural historically-acidic lakes.[18]

Changes in algae community structure induced by acidification may also alter zooplankton community structure. Acidification of lakes is accompanied by changes in abundance, diversity, and seasonality of zooplankton, which may reflect changes in their food base (algae), predators (fish) and/or complex changes in water chemistry. Since both population density and average size of the animals are reduced, food availability to fish and other animals may be reduced.[19]

Limited findings from New York State suggest that acid-tolerant algae may cover submerged aquatic plant communities in acidic lakes, thereby preventing them from receiving the sunlight necessary for growth.[20] Studies of Swedish lakes and preliminary information from New York, Nova Scotia, and Ontario have shown that acidification tends to cause decline of aquatic plants and replacement by growths of sphagnum mosses on lake and river bottoms.[21] Sphagnum moss creates a unique habitat which is considered unsuitable for some species of bottom-dwelling invertebrates or for use as fish spawning and nursery grounds.[22] Dense sphagnum beds may also reduce the appeal of freshwater lakes and rivers for recreational activities.

A number of studies have also found that acidic waters are more favorable to fungi than to normal bacterial populations. In many Scandanavian lakes studied to date, an increase in bottom accumulation of organic matter has been observed. This has been attributed to a shift in dominance from bacteria to fungi, which are less effective at decomposing organic material and which delay the recycling of nutrients.[23]

Table 1 and Figure 1 summarize the effects of decreasing pH on aquatic organisms. Table 1 describes biological processes affected in different organisms as the pH of water decreases. Figure 1 displays the percent of each

of 7 major categories (taxonomic groups) remaining as pH decreases. For example, at pH 7, 100% of normal mollusc species are present. As pH decreases, the number of species remaining decreases rapidly. At a pH of 5.5, all mollusc species have disappeared.

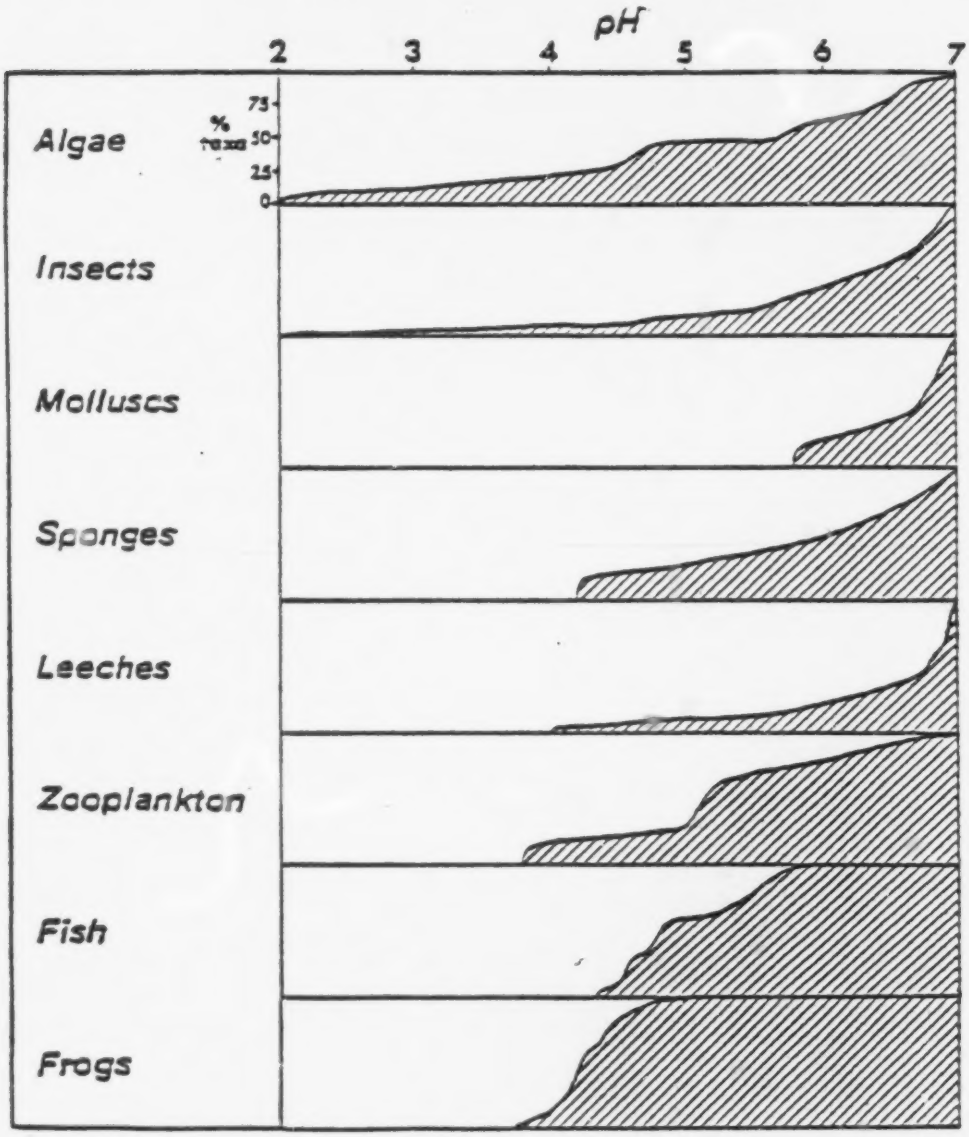
TABLE 1

EFFECTS OF DECREASING pH ON AQUATIC ORGANISMS

pH	Effect
8.0 - 6.0	In the long run, decreases of less than one-half of pH unit in the range of 8.0 to 6.0 are likely to alter the biotic composition of lakes and streams to some degree. However, the significance of these slight changes is not great. Decreases of one-half to one pH unit (a three to ten-fold increase in acidity) may detectably alter community composition. Productivity of competing organisms will vary. Some species will be eliminated.
6.0 - 5.5	Decreasing pH from 6.0 to 5.5 will reduce the number of species in lakes and streams. Among remaining species, significant alterations in the ability to withstand stress may occur. Reproduction of some salamander species is impaired.
5.5 - 5.0	Below pH 5.5, numbers and diversity of species will be reduced. Reproduction is impaired and many species will be eliminated. Crustacean zooplankton, phytoplankton, molluscs, amphipods, most mayfly species, and many stone fly species will begin to be eliminated. In contrast, several invertebrate species tolerant to low pH will become abundant. Overall, invertebrate biomass will be greatly reduced. Certain higher aquatic plants will be eliminated.
5.0 - 4.5	Below pH 5.0, decomposition of organic detritus will be impaired severly. Most fish species will be eliminated.
4.5 and below	In addition to exacerbation of the above changes, many forms of algae will not survive at a pH of less than 4.5.

Source: International Joint Commission, Great Lakes Advisory Board (1979), after Hendrey, 1979.

Figure 1: RELATIVE NUMBER OF TAXA OF THE MAJOR TAXONOMIC GROUPS AS A FUNCTION OF pH



SOURCE: Impact Assessment, U.S. - Canada Memorandum of Intent on Transboundary Air Pollution, Work Group I, Phase II Report, 1981.

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23. Bick, H., and E. F. Drews. 1973. Self purification and silicate communities in an acid milieu (model experiments). Hydrobiologia 42:393-402.

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J. HEALTH EFFECTS

1. SULFUR OXIDES

Summary

Overview

Approximately 30 million tons of sulfur dioxide (SO_2) are emitted each year in the United States from combustion of fossil fuels. Although SO_2 is harmful to human health in high concentrations (greater than 5220 micrograms(ug)/m^3, or 2.0 ppm), it is generally considered to pose no significant health effects at normal environmental concentrations (the primary standard for SO_2 is 365 ug/m^3 -- 0.14 ppm -- 24 hour average concentration). Greater concern attends sulfate particles -- e.g., sulfuric acid and ammonium sulfate -- that form through reactions between SO_2 and other substances in the atmosphere. Because these particles are extremely small (mostly under one micron in diameter), they can be transported over long distances in the atmosphere, and can readily be inhaled into the deep passages of the lung.

Acute exposures to sulfates (greater than 315 ug/m^3, or 0.08 ppm) constrict lung passages and lengthen lung clearance times in humans and laboratory animals; chronic exposure of laboratory animals to sulfuric acid mist produced evidence of the onset of chronic lung disease. In addition, numerous epidemiological studies have found correlations between ambient sulfate concentrations and mortality rates. Effects at current ambient levels (less than 23 ug/m^3, or 0.006 ppm) are highly uncertain.

Researchers at Brookhaven National Laboratory (under contract to OTA) have estimated that about 2% (a range of 0-5%) of the deaths per year in the U.S. and Canada might be attributable to atmospheric sulfur-particulate

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pollution. The range reflects uncertainties within the scientific community about the causal relationship between air pollution and mortality.

Sulfur Dioxide

When SO_2 is inhaled, it is normally absorbed by secretions lining the respiratory passage before it penetrates deeply into the lung. Sulfur dioxide emissions are currently regulated under National Ambient Air Quality Standards of the Clean Air Act; the primary standard for SO_2 is 365 ug/m^3 (0.14 ppm) (24-hour average concentration). Air pollution episodes characterized by particulates and very high levels of SO_2 have resulted in increased deaths in people with preexisting heart and lung disease. Changes in the function of the lungs have been seen in sensitive groups at concentrations above 5220 ug/m^3 (2.0 ppm). Recently, a study of asthmatics has shown that constriction of bronchial passages during periods of moderate exercise can occur at concentrations as low as 1300 and 260 ug/m^3 (0.5 and 0.1 ppm),[1] but these results have not yet been confirmed.

SO_2 Transformation to Sulfates

Once in the atmosphere, SO_2 is converted into sulfate particles at a rate of approximately 0.5 percent per hour.[2] The chemical reactions causing sulfate formation under atmospheric conditions are complex; much remains unknown about the pathways and interactions among the chemical components of

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1. Sheppard, D., A. Saisho, J. A. Nodel, and H. A. Boushey, 1981. "Exercise increases sulfur dioxide-indiced bronchoconstriction in asthmatic subjects. Am. Rev. Resp. Dis. 123:486.

2. Estimates of conversion rates under differing atmospheric conditions range from 0 to 4 percent per hour, on average, occasionally approaching 25% or greater per hour under high ambient oxidant conditions.

polluted atmospheres. Two main types of sulfate particles are known to be produced: sulfuric acid (H_2SO_4) and ammonium sulfate [$(NH_4)_2SO_4$]. Both types of particles are extremely small, and may be inhaled deep into the lung. The high acidity of sulfuric acid particles make them of primary concern for medical researchers, although ammonia in human breath may neutralize sulfuric acid; ammonium sulfate has not produced the pulmonary effects in animal and clinical studies that are seen with sulfuric acid mist.

Experimental Evidence of Sulfate-Related Health Damages

In laboratory experiments, acute exposures to high concentrations of sulfuric acid particles have a variety of adverse health effects. Concentrations of about 750 ug/m^3 (0.18 ppm) have irritated eyes and temporarily decreased vision; concentrations of 350 ug/m^3 (0.085 ppm) have increased breathing rates and altered lung function in asthmatics; changes in lung clearance rates have been observed in healthy nonsmokers at concentrations of 100 ug/m^3 (.025 ppm). Populations at special risk from particulate sulfates are the elderly and adults with preexisting chronic heart or lung disease. Children also appear to be especially susceptible to increased lower respiratory tract illness and decreased lung function. However, there is no direct evidence showing detectable effects on human health from maximum likely environmental concentrations of sulfate particles alone, although laboratory and chemical studies have verified that these small sulfate particles concentrate deeper in the lung than larger inhaled particles. In addition, there is evidence that acid sulfates (and SO_2) renders lung tissues more susceptible to the carcinogenic effects of polycyclic organic matter.

Evidence of Sulfate-Particulate Related Health Damages

Substantial evidence has been gathered over more than thirty years indicating injury from some aspect of the sulfate-particulate mix in air pollution. At high exposure levels, sulfur-particulate air pollution can aggravate asthma, chronic bronchitis, and heart disease.[3] There is also evidence that sulfur-particulate air pollution causes increased acute respiratory disease in children.

Many analyses of cities with different air pollution levels, comparing death rates among specific population groups, or cohorts, have shown strong correlations between pollution levels and mortality. Evidence is strongest for correlation between acute episodic effects and severe air-pollution incidents; evidence of long-term effects at comparatively low levels of pollution is more limited, and more controversial.

Nonetheless, scientists have generally been unable to attribute effects to any single element of the pollution mix. During the early 1970's, evidence seemed to point to sulfate particles as the health-damaging agent in the sulfate-particulate mix. More recent evidence suggests that the combination of sulfates and other associated particles such as metallic ions, nitrates, and fine soot particles may cause the observed effects.

Quantitative Estimates of Health Damages From Long-Range Transport Air Pollutants

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3. Environmental Protection Agency, "Review of the National Ambient Air Quality Standards for Particulate Matter: Draft Staff Paper," Strategies and Air Standards Division OAQPS/EPA/RTP June 13, 1981, October 1981, January 1982; Lave, Lester B. and E. P. Seskin, Air Pollution and Human Health, 1977, John Hopkins University Press, Baltimore, Maryland; Perera, Frederica and A. K. Ahmed, Respirable Particles: Impact of Airborne Fine Particulates on Health and the Environment, 1979, Ballinger Publishing Company.

To derive quantitative estimates of damage caused by long-range pollution transport, the contractor (Brookhaven National Laboratory) has used sulfate concentrations as an index of the sulfur-particulate air pollution mix, acknowledging, "Its use in this manner remains controversial, but it is the best we can do under the current state of knowledge." The contractor also used a previously-developed health-damage function which specifically addresses uncertainties in knowledge by projecting a range of mortality estimates for a given population exposure level.[4]

The health-damage function used is essentially a compilation of expert opinion about the relationship between sulfate pollution and premature mortality (e.g., due to aggravation of pre-existing respiratory or cardiac problems). The range reflects a controversy over the validity of epidemiological (i.e., statistical) studies indicating a relationship between mortality and air pollution levels, and includes estimates from scientists who believe the relationship is reasonable as well as from those who do not.

These studies, carried out and extensively examined over the past decade, examine mortality statistics from areas of the country exposed to different levels of air pollution. The value of these studies is that they examine human health directly, without the necessity of extrapolating from animal studies or small samples of people. They indicate that there are regional differences in mortality, and that regional patterns of mortality are similar to air pollution patterns. While it is possible that factors not considered in these analyses may be more important than the sulfur-particulate mix, none

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4. The function, developed by Morgan et al., is a probabilistic one with a 90 percent confidence interval of 0-11 excess deaths per 100,000 person-ug/m^3 sulfate exposure (i.e., deaths that would not occur in the absence of sulfate). This function is currently under revision.

have been demonstrated to date. Such studies are most useful for examining the potential effects of chronic, low-level exposure -- what might be happening and what might result from future air pollution levels.

Because the mortality estimates developed for OTA assume that there is no damage threshold (i.e., minimum exposure level at which air pollution begins to affect mortality), these estimates must be considered an upper bound. The available medical evidence neither confirms nor refutes the existence of such a threshold.

Sulfate concentration data were derived from SO_2 emission inventories developed for the OTA study, using the Regional Climatological Dispersion Model (RCDM-2), developed at the University of Illinois and currently under evaluation by the U.S.-Canadian Work Group on Transboundary Air Pollution. Population estimates and projections were derived from the U.S. Census Bureau and Canadian Ministry of Industry, Trade, and Commerce data.

Estimates of excess deaths due to sulfate-particulate air pollution were made for: (1) 1980 population levels using 1978 SO_2 emissions data; (2) populations in the year 2000, assuming that total SO_2 emission levels are held constant at 1978 levels; and (3) populations in the year 2000, assuming that SO_2 emission levels are reduced by 30 percent from 1978 levels. Results of these three scenarios are presented in Table 1.

Table 1
Summary of Scenario Results

SCENARIO	Population (millions)	SO_2 Emissions (10^3 tons)	Excess Deaths Total (10^3 deaths)	Range	Rate (Deaths/10^5) Population)
1. 1980 Population, 1978 Total SO_2 Emissions, U.S. and Canada	249	34.	51.	0-150.	20.4
2. 2000 Population, SO_2 Emissions unchanged, U.S. and Canada	291	34.	57.	0-170.	19.8
3. 2000 Population SO_2 emissions 30% below 1978 Emissions, U.S. and Canada	291	24.	40.	0-120.	13.8

Total SO_2 emissions in the U.S. and Canada have been estimated to result in 51,000 premature deaths under the first scenario, increasing 1978 mortality rates by about 2 percent -- an amount approximately equivalent to the number of deaths attributable to infectious lung diseases. While increase in population by the year 2000 would cause 1978 emission levels to induce slightly higher levels of premature deaths (Scenario 2), a 30 percent decrease in overall SO_2 emissions by 2000 is estimated to reduce the number of deaths annually attributable to air pollution to 40,000, or 1.6 percent of the total mortality rate. Estimates of the total exposure exported across the U.S.-Canadian border suggests that each country "exports" approximately

2,000-3,000 excess deaths per year.[5]

Estimating mortality attributable to pollution exposure does not necessarily imply that the mortality impacts are the most significant. Mortality dose-response functions are used because the available data on morbidity are inadequate for detailed epidemiological analyses. It has been estimated that, for every person dying from sulfate exposure, five persons are ill with lung disease, heart disease aggravated by lung dysfunctions, and respiratory infections aggravated by lung irritations.[6]

2. NITROGEN OXIDES

The term nitrogen oxides (NO_x) refers to a number of compounds -- NO, NO_2 and such secondary byproducts as nitric acid, nitrate aerosols, and nitrosamines[7] -- all of which have the potential to affect human health. The principal target of nitrogen oxides pollution is the respiratory system; of the several NO_x compounds, by far the greatest amount is known about the effects of NO_2, a pollutant which has been studied over the past 30 years and is presently regulated under national ambient air quality standards.

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5. The estimated total exposure exported from the U.S. to Canada is 94 million person-ug/m^3; total exposure imported from Canada to the U.S. is 62 million person-ug/m^3. The proportional contribution of each country's exports to the other's attributed excess deaths is somewhat different, because of differences in internally-produced sulfate levels and in population densities.

6. Hamilton, L. D. 1980. Comparative risks from different energy systems: Evolution of the Methods of Study." Societe Francaise D'Energie Nucleaire, Colloquium on the Risks of Different Energy Sources. Paris, France, January 24-26, 1980.

7. These secondary byproducts are produced from photochemical transformation of NO and NO_2 in the atmosphere; nitrosamine formation has been theoretically suggested, but not conclusively proven.

Episodic or peaking concentrations of NO_2 (possibly augmented by NO) have been shown to cause immediate and short-term irritation to such sensitive subgroups of the population as asthmatics and individuals recovering from acute respiratory infections. Chronic exposure to lower concentrations of NO_2 has been associated with increased occurrence of acute respiratory infections in infants and children. Some scientists have hypothesized that such repeated low-level exposures are also associated with chronic lung disease and increased "aging" of the lung in adolescence and adulthood.

Health effects directly attributable to NO_2 are generally considered to result from localized sources of pollution. Combustion of coal, oil, and gasoline is estimated to be responsible for local outdoor concentrations of NO_x 10-100 times naturally-occurring background levels in areas of high emissions.[8] Indoor sources of NO_x, such as gas cooking stoves, some home heating systems, and cigarette smoking, are also known to contribute to population exposure, and in some cases may exceed outdoor concentrations of NO_x.

Byproducts of nitrogen oxides are known to be transported over long distances, potentially contributing to acid deposition and ozone formation. Some of these secondary pollutants are more toxic on a per-unit basis than NO_2; however, it is not currently known whether they are present in sufficient concentrations -- or persist long enough -- in the atmosphere to endanger human health. Recent studies performed for the EPA and the National Academy of Sciences suggest that nitrosamines and similar chemical species may be

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8. Mobile-source combustion was estimated to contribute 44 percent of anthropogenically-produced NO_x for 1976 in the U.S., while stationary-source combustion accounted for 56 percent. USEPA, Air Quality Criteria for Oxides of Nitrogen, June 1979, pp. 1-2 and 1-3.

found in sufficient concentrations in polluted urban air near certain industrial areas to warrant concerns for health. Not enough is yet known about the effects of nitrogen oxides, or their concentrations in the atmosphere, to permit quantitative estimation of the health-related damages for which NO_x may be responsible, or of the population at risk from these pollutants.

Summary of Medical Research

Clinical studies of the short-term effects of NO_x on human volunteers have been conducted on asthmatics, patients with bronchitis, and healthy subjects. The studies have generally shown that the sensitivity of asthmatics to irritants such as cold air or air pollutants can be heightened by short-term concentrations of NO_2 as low as 940 ug/m^3 (0.5 ppm); the primary National Ambient Air Quality standard for NO_2 is 100 ug/m^3 or 0.05 ppm. Recovery from these effects tends to be rapid, and it is not known whether repeated exposures of this kind have any cumulative effects, or predispose the lungs to permanent damage. In general, heightening of sensitivity or irritation has not been shown for healthy subjects or bronchitis patients at NO_2 concentrations at or below 2820 ug/m^3 (1.5 ppm). One study reports healthy adults to be unaffected by acute exposures of up to 9400 ug/m^3 (5 ppm), while another study found sensitivity in healthy adults to be heightened by concentrations as low as 94 ug/m^3 (0.05 ppm) NO_2.

Laboratory experiments have tested the reactions of animal tissues to similar and higher concentrations of NO_2. These experiments have provided significant insight into the mechanisms which govern human reactions to NO_2 exposures, and suggest more serious consequences of long-term human exposure to NO_2 than have appeared in short-term clinical studies.

Prolonged exposure to NO_2 has been observed to cause damage to lung tissue in laboratory animals. The principal consequences of such damage appear to be development of emphysema-like conditions, and reductions in resistance to respiratory infection. For a fixed dosage, greater concentrations have been shown to cause greater increases in mortality rates for animals exposed to respiratory infection than greater duration of exposure. This suggests that fluctuating levels of NO_2, such as are found in community air, may prove more toxic than sustained levels of the gas.

Epidemiological studies of populations of children exposed to NO_2 concentrations, primarily via indoor air pollution, confirm laboratory findings of reduced resistance to respiratory infection in exposed animals. Children exposed to additional quantities of NO_2 (and possibly NO) from gas stove combustion -- in particular, infants under the age of two -- show significantly greater incidence of acute respiratory illness and changes in lung function than counterparts living in homes with electric stoves. Similar findings have not been observed in adults.

3. INDIRECT HEALTH EFFECTS OF ACID PRECIPITATION

Overview

Acidified waters are known to be capable of dissolving toxic metals -- e.g., aluminum, copper, lead, and mercury -- and releasing such toxic substances as asbestos, from soils and rocks in watersheds and lakes, and from drinking water distribution systems. Researchers are attempting to determine the extent to which the total body burden of these substances in humans might result from acidic deposition, as opposed to direct inhalation or occupational exposure.

No direct relationship has yet been established between acid deposition and degradation of drinking water quality. Potentially harmful levels of asbestos and lead have, however, been found in "aggressive" (i.e., corrosive) acidified waters in the eastern U.S. Scientists have encountered difficulty in pinpointing the direct source of concentrations of toxic materials found in acidic tap waters. Acidic precipitation can scavenge toxic materials from the atmosphere during rainfall events, leach them from soils and rocks as they pass through the watershed, or leach them from pipes and conduits used to distribute water to users.

The aggressiveness of municipal water supplies can be monitored and corrected fairly easily. A recent amendment to the National Interim Primary Drinking Water Regulations has established the initial steps in a corrosion control program.[9] Public water systems are now required to identify the presence of specific materials (e.g., lead and cadmium) within the distribution system and to monitor corrosiveness characteristics at least one year. Also, the types of materials used in the distribution system and home plumbing must be reported. Anti-corrosion treatment is currently being practiced in specific areas (e.g., Boston) where acid precipitation is known to affect water quality. However, aggressive well water in rural areas where soils have little capacity to counteract acid deposition is more difficult to detect and mitigate, and potential health effects remain of concern.

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9. Environmental Protection Agency. Interim Primary Drinking Water Regulations; Amendment. Final Rule, Fed. Reg. 45(168): 57332-57357, 27 August 1980a.

Asbestos

Asbestos occurs widely in natural rock formations throughout the east and west coasts of the U.S. In addition, asbestos fibers are mixed in concentrations of 10 to 25 percent with cement to reinforce pipe used to distribute water supplies. While studies have shown that "water can corrode asbestos-cement materials and release fibers into the drinking water", data showing a direct association between precipitation pH and asbestos content of drinking water are not available. However, preliminary estimates of asbestos in drinking water from asbestos/cement distribution pipes suggest that approximately 11 percent of the total U.S. population is exposed to asbestos concentrations greater than 10 million fibers per liter.[10]

Correlations between above-average incidence of certain abdominal cancers and elevated concentrations of asbestos in drinking water have recently been found in the San Francisco Bay area, where abundant natural supplies of asbestos occur in bedrock.[11] EPA has used data from studies of workers exposed to airborne asbestos to estimate that one additional cancer will be caused for every 100,000 people exposed to drinking water containing 300,000 asbestos fibers per liter for a 70 years.

Lead

Lead enters aquatic ecosystems from wet and dry precipitation, mineral

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10. Millette, J. R., M. F. Pansing, and R. L. Boone. Asbestos-cement materials used in water supply. Water Engineering and Management, 128:48, 51, 60, 97, 1981a.

11. Kanarek, M. S., P. M. Conforti, L. A. Jackson, R. C. Coper, and J. C. Murchio. Asbestos in drinking water and cancer incidence in the San Francisco Bay area. Am. J. Epidemiol., 112:54-72, 1980.

erosion, street runoff, leaded gasoline, and municipal and industrial discharge, as well as from corrosion of lead in pipes and plumbing systems. Corrosion of lead from lead-containing materials is facilitated by waters of low pH (less than 6.5) -- especially when the waters are also of low alkalinity. Individual water samples in isolated areas characterized by low pH values have shown lead concentrations of up to one hundred times the 50 microgram ambient water quality standard.[12]

While elevations in human blood lead levels will result directly from ingesting contaminated drinking water, the human body also absorbs lead from a number of other sources, including food, ambient air, cigarettes, paint, dust, and dirt.[13]

Mercury

Studies in Scandanavia, Canada, and the U.S. have found correlations between elevated levels of mercury in fish and increasing acidity of lake and stream habitats. However, no cases of mercury poisoning are known to be

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12. Mushak, P. Multimedia pollutants. Report to the National Commission on Air Quality. Contract No. 232-AQ-6981, 1981; Fuhs, G. W. and R. A. Olsen. Technical memorandum: Acid precipitation effects on drinking water in the Adirondack Mountains of New York State. Dept. of Health, New York State, Albany, N.Y., 1979.

13. EPA has defined approximately 2,500,000 people -- inner-city children under 5 and pregnant women -- as being at special risk from elevated levels of lead in the bloodstream. While aggressive drinking waters can leach lead from pipes and contribute to lead levels in the blood, such effects may constitute a greater threat to rural populations using acidified groundwaters than to urban populations using waters that can readily be treated. Preliminary investigations in the New York State Adirondacks region have found isolated cases of human exposure to lead-contaminated drinking water resulting in lead concentrations in blood as high as 59 ug/dl. (New York State Dept. of Health, Technical Memorandum: "Lead Poisoning Related to Private Drinking Water Sources," October 23, 1981.)

associated with freshwater fish in the U.S. Lakes in acid-sensitive regions of Minnesota, Wisconsin and in the Adirondacks of New York have yielded fish with mercury concentrations above the FDA public health standard of 0.5 ppm[14,15]. Studies are currently underway to assess correlations between observed mercury levels and lake pH levels.

A number of recent efforts to neutralize acidic lakes through the application of lime appear to be successful in reducing mercury levels in fish. Favorable results have been reported for various locations in Sweden, and for Nelson Lake in Ontario, Canada.[16]

Other Metals

Acidified waters are known to leach substantial amounts of aluminum from watersheds, and the solubility of aluminum increases as pH decreases below 6.0. Concentrations of aluminum in acidified well-water in the U.S. have been found as high as 1.7 mg/l, which could represent a substantial portion of an individual's daily aluminum intake. Little is known about the toxicity of aluminum to most human populations; no restrictions have been established in the U.S. on aluminum concentrations in drinking waters or foods.

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14. Acid Precipitation in Minnesota, Minnesota Pollution Control Agency, Minnesota Dept. of Natural Resources, Minnesota Department of Health, January 26, 1982, p. 7-67; Letter from G. Wolfgang Fuhs, Director, N.Y.S. Dept. of Health, to Congressman Toby Moffett, July 16, 1980, p. 3.

15. Personal communication, James Wiener, U.S. Fish and Wildlife Service, La Crosse Field Research Station, La Crosse, Wisconsin, May 1982.

16. "The Role of Clouds in Atmospheric Transport of Mercury and Other Pollutants," G. H. Tomlinson, R. J. P. Brouzes, R. A. N. McLean, and John Kadlecek, in Ecological Impact of Acid Precipitation, Proceedings of an International Conference, Sandefjiord, Norway, March 11-14, 1980, Oslo-As.

Several studies have shown, however, that elevated aluminum concentrations may be toxic to individuals with impaired kidney function -- e.g., water containing more than 50 ug/l of aluminum is thought to be unsafe for dialysis treatment.

For normal human beings, copper deficiency is more widespread than copper toxicity. A limit of 1 mg/l for copper concentration in U.S. drinking water has been set on the basis of taste; no regulatory limit exists for copper concentrations in food. Acidic waters have been shown to be capable of corroding copper from pipes in household distribution systems -- copper levels in several drinking water samples from sensitive areas of the Adirondacks exceeded the 1 mg/l standard[17]. While several cases of copper toxicity have been reported in the U.S. among individuals with Wilson's disease, a once-fatal disorder of copper metabolism, the hazard to the general population from currently-observed copper levels appears quite small.

In its natural state, cadmium occurs as an impurity in zinc, copper-zinc, or lead-zinc deposits. In concentrated or pure form it is very toxic. Little evidence of cadmium corrosion from galvanized pipe or plumbing alloys currently exists, although studies of municipal water systems have shown occasional exceedances of the U.S. health standard of 10 ug/l. The concentrations found in drinking water, however, represent only a minute fraction of the amounts known to cause acute cadmium poisoning or chronic health effects under industrial exposure conditions. Based on the infrequent occurrence of cadmium in drinking water at levels above 10 ug/l, such exposures are not considered a threat to many people.

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17. Fuhs and Olson, 1979.

K. ECONOMIC SECTORS INVOLVING RESOURCES CONSIDERED SENSITIVE TO THE EFFECTS OF TRANSPORTED POLLUTANTS

Certain sectors of the U.S. economy -- in particular, farming, forestry, and the recreational industry associated with freshwater fishing -- are among the most vulnerable to potential effects of transported air pollutants. However, current levels of understanding about the resource damages that acid deposition and ozone may cause to different sectors vary substantially. In a previous section, quantitative estimates of the magnitude of crop losses due to ozone were presented by region and crop. For forests, only areas potentially susceptible to damages, rather than estimates of current or future damage, were presented. Estimates were also developed for the number of lakes and miles of streams either acidified or undergoing alteration due to acid deposition, but these regional estimates may not correspond to the types of lakes and streams most often used for fishing or other recreational activities.

To examine the economic implications of potential pollution-related losses, information about the magnitude and geographic distribution of the uses of these resources is desirable. For this purpose, OTA has assembled data bases from the Department of Labor and the U.S. Fish and Wildlife Service to estimate levels of activity in agriculture, forestry, and freshwater fishing for 1980 in the 31-state eastern U.S. region. These tabulations reflect the importance of each of these sectors, but cannot yet be used to estimate the number of people that might be affected by changes due to long-range transported pollutants without additional information.

The state-by-state estimates provided in Tables 1 and 2 calculate the number of people employed in agriculture and forestry, the amount of wages, salaries, and proprietary income paid to them, and the proportions of total state employment and income that these figures represent. In the ARMS region overall, agriculture and related services provided employment for approximately 3 million people, or 4 percent of total employment, and directly produced slightly less than $35 billion, or 3.1 percent of the region's total income. As would be expected, agriculture and related services account for ten percent or more of employment in Iowa, Arkansas, and Kentucky, but under one percent of employment for Massachusetts, Rhode Island, and New Jersey. Agricultural enterprise does not, however, follow strict regional patterns -- in Vermont, for example, it employs 6.1 percent of the workforce, and produces 7.8 percent of workforce income, while in Illinois agriculture accounts for only 3.1 percent of employment and 2.9 percent of income.

Forestry is estimated to have employed about one million people in 1980, or 1.5 percent of the region's total, and produced slightly less than $20 billion, or 1.8 percent, of regional income. State-to-state variations are significant; three states -- Maine, Wisconsin, and Arkansas -- employ three percent or more of the workforce in forestry, while nine other states employ less than one percent. In general, forestry provides higher proportions of state income than employment throughout the ARMS region -- at least 50 percent higher for such states as Connecticut, Maine, Wisconsin, Minnesota, and South Carolina.

The 1980 National Survey of Fishing and Hunting conducted by the Fish and Wildlife Service is the basis for estimates in Table 3 of the number of participants in, and the level of their expenditures for, freshwater fishing

in the 31-state region. In total, over 21 million people are estimated to have spent approximately $6.3 billion on fishing-related activities in lakes and streams in the ARMS region in 1980. Expenditure estimates are included for travel, food, lodging, fees, and light equipment used exclusively for fishing. Nine states are estimated to have had over a million in-state residents participating in fishing activities; fourteen states had over a million participants when out-of-state fishermen are included. In the states of Florida and Wisconsin, where fishing is an important tourist attraction, visitors from outside the state spent over $100 million on fishing-related activities; out-of-state residents also spent slightly under $100 million in Minnesota, Arkansas, Missouri, and Michigan. As shown in Table 3, recreational freshwater fishing is an activity generating at least one-quarter of a billion dollars annually for each of twelve states in the region.

TABLE 1

1980	Employment in Farms & Agricultural Services[1] No. of People	% of State Total	Income from Farms & Agricultural Services[1] Thousands of $	% of State Total
U.S. TOTAL	4,628,300	4.3	59,195,000	3.7
NEW ENGLAND	96,547	1.6	889,035	1.0
Connecticut	20,436 [3]	1.3	166,349 [3]	.6
Maine	23,069	4.4	185,254	2.9
Massachusetts	25,670 [3]	.9	203,147 [3]	.5
New Hampshire	7,987	1.8	35,519	.6
Rhode Island	3,121	.7	24,779	.4
Vermont	15,220	6.1	232,113	7.9
MIDEAST	312,104	1.6	3,750,607	1.2
Delaware	7,116	2.4	179,116	3.7
District of Columbia	161	.0	3,216	.0
Maryland	38,293	1.9	417,505	1.4
New Jersey	30,570	.9	347,551	.6
New York	117,351	1.5	1,187,158	.9
Pennsylvania	118,613	2.2	1,616,061	2.0
GREAT LAKES	737,196	3.9	9,526,576	3.1
Illinois	168,380	3.1	2,624,316	2.9
Indiana	134,314	5.3	1,646,415	4.4
Michigan	116,823	3.1	1,252,389	1.8
Ohio	154,279	3.2	1,499,706	1.9
Wisconsin	163,400	7.2	2,503,750	7.7
PLAINS REGION				
Iowa	177,130	12.3	2,114,886	11.1
Minnesota	161,453	7.6	2,452,055	8.1
Missouri	165,696	7.2	2,503,750	7.7
SOUTHEAST	1,360,759	5.7	13,716,756	4.3
Alabama	101,110	6.0	986,742	4.5
Florida	135,023	3.2	2,869,736	4.9
Arkansas	103,621	10.6	1,558,117	13.2
Georgia	119,319	4.6	1,172,888	3.4
Kentucky	154,212	10.0	1,186,024	5.7
Louisiana	72,610	3.9	791,605	2.9
Mississippi	104,257	9.6	1,006,862	8.1
North Carolina	177,556	6.1	2,374,681	6.5
South Carolina	76,427	5.1	466,992	2.7
Tennessee	144,397	6.7	686,352	2.5
Virginia	100,666	3.8	529,633	1.5
West Virginia	30,607	4.1	87,085	.8
31-State Total	3,010,885 (65%)	4.0	34,762,914 (59%)	3.1

1. Includes farm proprietors, excludes manufacture and sale of farm equipment.

2. Includes construction of prefabricated buildings and mobile homes; excludes manufacture of furniture, printing and publishing industries, and sale of building materials.

3. 1980 data not available for all data points -- estimates may be slightly low due to use of data from mid-1970's.

TABLE 2

1980	Employment in Forestry, Lumber Wood and Paper Products[2] No. of People	% of State Total	Income from Forestry, Lumber, Wood and Paper Products[2] Thousands of $	% of State Total
U.S. TOTAL	1,404,000	1.0	27,717,000	1.7
NEW ENGLAND	98,846	1.6	1,962,320	2.2
Connecticut	11,872	.7	281,737	1.1
Maine	32,188	6.2	667,927	10.6
Massachusetts	33,395 [3]	1.2	647,318 [3]	1.5
New Hampshire	11,066 [3]	2.5	202,203 [3]	3.5
Rhode Island	3,901	.9	59,024	.9
Vermont	6,409	2.6	53,647	1.8
MIDEAST	184,360	.9	3,674,985	1.1
Delaware	4,082 [3]	1.4	54,943 [3]	1.1
District of Columbia	384 [3]	.1	6,786 [3]	.1
Maryland	13,824	.7	247,515	.8
New Jersey	38,599	1.1	795,801	1.4
New York	64,965	.8	1,292,723	1.0
Pennsylvania	63,580	1.2	1,266,879	1.5
GREAT LAKES	233,338	1.2	4,691,795	1.5
Illinois	47,079	.9	916,609	1.0
Indiana	32,791	1.3	567,832	1.5
Michigan	32,791	.8	682,294	1.0
Ohio	52,404	1.1	1,060,559	1.4
Wisconsin	68,842	3.0	1,474,502	4.5
PLAINS REGION				
Iowa	8,464	.6	155,655	.8
Minnesota	43,415	2.0	1,054,100	3.5
Missouri	23,910	1.0	399,616	1.2
SOUTHEAST	428,391	1.8	7,688,675	2.4
Alabama	49,056	2.9	925,328	4.2
Florida	39,191	.9	719,840	1.2
Arkansas	34,104	3.5	612,986	5.2
Georgia	59,375	2.3	1,123,053	3.2
Kentucky	16,749	1.1	284,560	1.4
Louisiana	28,516	1.5	594,700	2.1
Mississippi	31,218	2.9	531,380	4.3
North Carolina	57,191	2.0	962,031	2.7
South Carolina	30,150	2.0	544,293	3.1
Tennessee	37,106	1.7	615,946	2.3
Virginia	38,351	1.4	660,070	1.8
West Virginia	7,350	1.0	101,506	.9
31-State Total	1,020,724 (73%)	1.4	19,728,147 (71%)	1.8

1. Includes farm proprietors, excludes manufacture and sale of farm equipment.

2. Includes construction of prefabricated buildings and mobile homes; excludes manufacture of furniture, printing and publishing industries, and sale of building materials.

3. 1980 data not available for all data points -- estimates may be slightly low due to use of data from mid-1970's.

TABLE 3

	NUMBERS OF STATE RESIDENTS WHO PARTICIPATED IN FRESHWATER FISHING - 1980[1]	TOTAL NUMBER OF PARTICIPANTS IN FRESHWATER FISHING - 1980[1]	AVERAGE NUMBER OF DAYS OF FISHING PARTICIPATION	FISHING RELATED EXPENDITURES (Travel, Lodging, Food, Fees, Equipment[3]) in Millions of Dollars		
				BY STATE RESIDENTS	BY OUT-OF-STATE RESIDENTS	TOTAL
ALABAMA	874,952	1,117,099	19	228.4	57.4	285.8
ARKANSAS	574,351	895,619	17	149.9	88.0	237.9
CONNECTICUT	271,488	307,217	18	59.2	2.1	61.3
DELAWARE	30,221	35,136	21	4.4	3.4	7.8
DIST.OF COLUMBIA	10,361	15,006	35	3.2	.5	3.7
FLORIDA	1,193,583	1,550,406	21	364.0	119.9	483.9
GEORGIA	1,117,544	1,253,265	19	274.9	25.9	300.8
ILLINOIS	1,264,434	1,401,922	18	297.1	24.4	321.5
INDIANA	913,212	1,111,095	20	209.1	59.6	268.7
IOWA	592,242	646,180	17	107.2	4.4	111.6
KENTUCKY	734,630	1,052,587	16	185.9	49.9	235.8
LOUISIANA	752,110	955,567	18	184.3	40.0	224.3
MAINE	214,505	364,062	14	39.9	43.7	83.6
MARYLAND	268,716	321,687	11	55.1	6.0	61.1
MASSACHUSETTS	333,680	414,012	18	78.7	7.6	86.3
MICHIGAN	1,292,682	1,666,295	17	320.6	74.3	394.9
MINNESOTA	1,175,026	1,639,131	13	285.6	97.9	383.5
MISSISSIPPI	538,349	689,901	20	144.8	25.5	170.3
MISSOURI	998,667	1,230,423	16	223.7	77.9	301.6
NEW HAMPSHIRE	115,357	224,998	14	18.4	17.8	36.2
NEW JERSEY	299,175	383,375	19	80.6	18.6	99.2
NEW YORK	1,036,257	1,225,313	13	208.3	32.5	240.8
NORTH CAROLINA	927,341	1,096,338	17	183.6	25.7	209.3
OHIO	1,412,022	1,500,525	21	269.7	10.5	280.2
PENNSYLVANIA	1,168,907	1,400,689	19	229.1	43.6	272.7
RHODE ISLAND	60,143	72,633	19	10.0	2.0	12.0
SOUTH CAROLINA	490,089	639,259	19	106.8	28.9	135.7
TENNESSEE	903,633	1,161,467	17	216.9	51.6	268.5
VERMONT	113,256	182,753	15	17.8	14.3	32.1
VIRGINIA	637,681	823,369	15	149.2	24.9	174.1
WEST VIRGINIA	341,293	416,874	19	91.1	9.4	100.5
WISCONSIN	1,109,738	1,674,219	16	278.5	162.0	440.5
31-STATE TOTAL	21,760,645	[2]		5,076.0	1,250.2	6,326.2

1. Does not include fishing in the Great Lakes.
2. Total not calculated due to potential double-counting of participants.
3. Equipment expenditures calculated for state residents only to avoid double-counting.

L. SOURCE-RECEPTOR RELATIONSHIPS

I. INTRODUCTION:

Broad regions of North America receive acidic deposition in both wet form -- acid rain -- and as dry deposition of acidic substances. Because acidifying substances reach the earth from the atmosphere both by being removed by rainfall and as gases and particles deposited directly from the air, the term acid rain is misleading. The acidity of rainfall per se is generally considered to be of less significance than the "acid-producing" substances* added to the environment -- onto soils, vegetation, and materials, and, after passing through watersheds, into lakes and streams.

The chain of physical and biological processes from emissions of pollutants to eventual deposition of acid-producing substances in the environment is complex and not fully understood. However, several lines of evidence can be combined to express the likely relationship between emissions and deposition.

Acid deposition originates from both local and distant sources of sulfur and nitrogen oxides. Given the current scientific understanding, one would expect that reductions in sulfur dioxide emissions throughout the 31-state eastern U.S. region would result in reduced deposition of acid-producing substances; but that this will occur in the areas sensitive to acid deposition cannot be stated with certainty. As will be discussed in this section, one

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* The term acid-producing substances as used in this report refers to sulfur and nitrogen oxides and other substances that have the potential for producing acidity, but may not be deposited in an acid form. For example, sulfate can be deposited in a neutral form -- ammonium sulfate -- but end up as sulfuric acid by the time it reaches a lake or stream if the ammonium is used by plants in the watershed.

might also say that, for the eastern U.S., no other pollutant offers a greater chance of achieving acid deposition reductions than sulfur dioxide emissions reductions. This does not mean that reductions in other pollutants could not be considered simultaneously, or might not be ultimately necessary to achieve a desired level of deposition reduction, but sulfur dioxide reductions would be the most likely initial focus of an attempt to control acid deposition.

By considering four key points with preliminary information drawn from several alternative approaches, one can piece together a plausible relationship between emissions of pollutants and deposition of acid-producing substances. The discussion will begin with the deposition of acidifying substances regionally and then, after identifying the major constituents of deposition, consider the relative magnitude of current sources. A series of model-based estimates of the effects of pollutant emissions in parts of the eastern U.S. on deposition in other areas is presented to illustrate the regional nature of acidic deposition. Two complimentary modeling approaches are used to estimate the magnitude of potential reductions of acid-producing substances reaching the environment due to reductions in sulfur dioxide emissions. Finally, by reconsidering current deposition levels, the analysis addresses the question of whether potential reductions in acid-producing substances estimated by these modeling exercises might be enough to meet "target" deposition rates to protect sensitive resources.

II. CURRENT DEPOSITION OF ACID-PRODUCING SUBSTANCES:

The best information on patterns of acidic deposition -- both the spatial distribution and the relative proportions of acid-producing substances -- comes from monitoring networks collecting rainfall samples. Though wet

deposition may account for only about half of the total deposition*, monitoring of dry deposition of gaseous and particulate pollutants is too limited to allow one to draw meaningful conclusions about relative proportions of pollutants.

Acidity of rainfall is caused by the balance of chemicals found in rain originating from natural sources -- such as sea-spray, windblown soil, and carbon dioxide in the air -- and man-made sources such as sulfur and nitrogen oxide pollutants. The major "acid producing" substances in rainfall in the eastern United States are from sulfur and nitrogen oxides from both natural and man-made sources. However, the presence of these substances does not necessarily indicate the presence of acid in rainfall; they can be counter-balanced by acid-neutralizing substances such as calcium and magnesium from soil and by ammonium from natural sources and fertilizers. The acidity of rainfall can be decreased by either removing acid-producing substances, or increasing acid-neutralizing substances.

The spatial patterns of deposition of acidity, sulfate and nitrate in rainfall over eastern North America are displayed in figures 1-3. Figure 1 presents the deposition of hydrogen ions -- the substance actually measured to determine acidity. Broad regions of the eastern U.S. receive wet deposition of hydrogen ions in excess of 40 milliequivalents per square meter per year

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* Most long-range transport air pollution models estimate that dry deposition is about equal to wet deposition when averaged over the eastern U.S. The ratio of dry to wet deposition varies with distance away from sources of pollution, with ratios on the order of 8 to 12 in areas with high SO_2 concentrations (D. Fowler, "Removal of Sulphur and Nitrogen Compounds from the Atmosphere", Ecological Impact of Acid Precipitation, SNSF project, October 1980) to about .2 to .4 in the Adirondack Mountains (A. H. Johnson et al. "Relationships between Wet Deposition, Dry Deposition and Through-fall Chemistry". Air Pollution Control Association Annual Meeting, New Orleans, June 1982.)

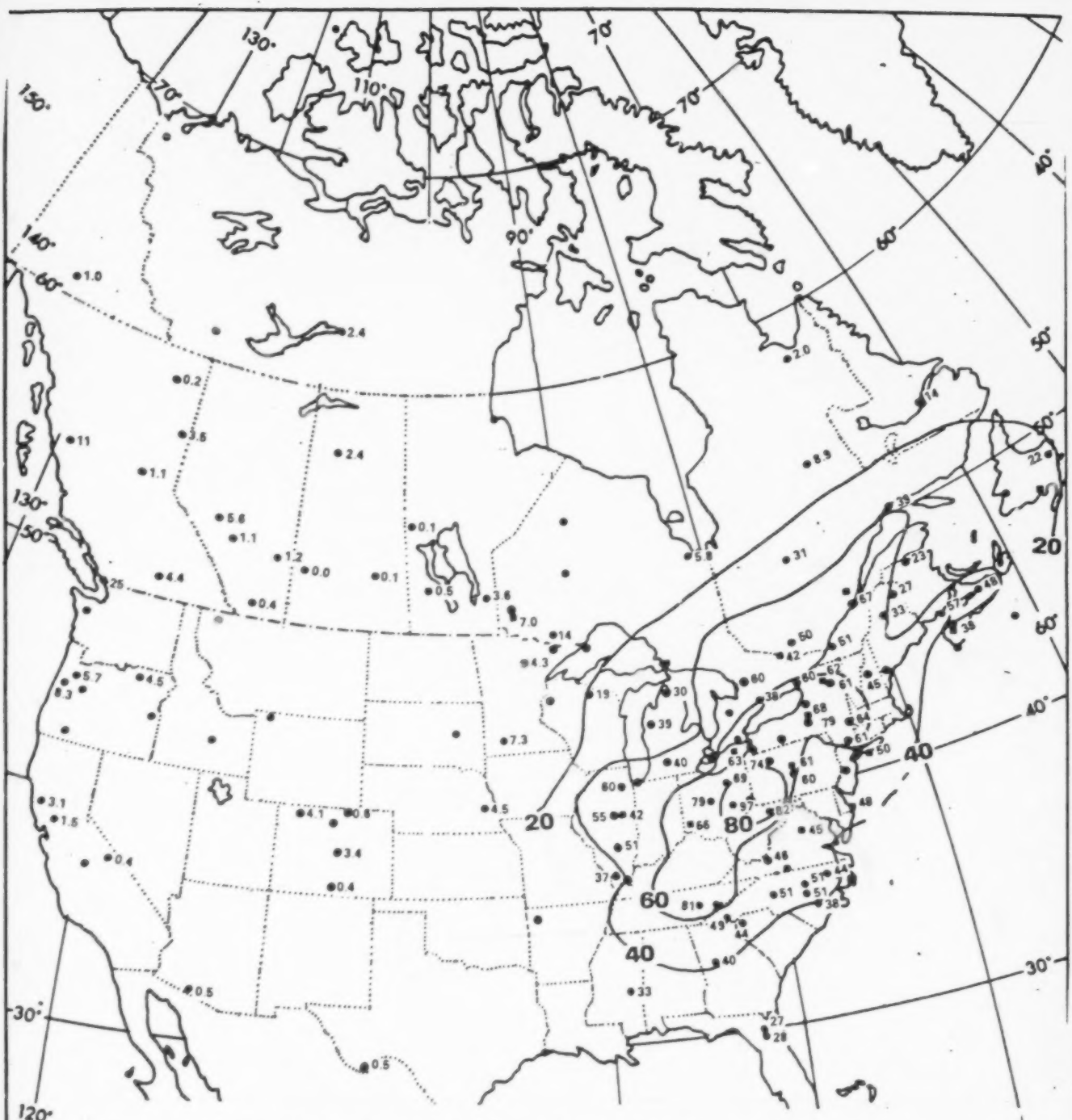


Figure 1. Spatial Variation of Precipitation-Amount-Weighted Mean Hydrogen Ion Deposition (milliequivalents per meter^2) in North America in 1980 Compiled from Canadian (CANSAP ●, APN ■) and American (NADP ●, MAP3S ■) Networks

(meq/m^2-yr). The highest deposition rates are in excess of 80 meq/m^2-yr, centered around eastern Ohio, western Pennsylvania and northern West Virginia.

Of the major acid-producing substances in rainfall, only two originate in substantial amounts from man-made sources in the eastern U.S.: sulfates and nitrates. Comparing the deposition of acidity to the deposition of sulfate (figure 2), the patterns are quite similar. A larger part of the eastern U.S. receives deposition in excess of 40 meq/m^2-yr. The highest deposition rates are again in excess of 80 meq/m^2-yr, centered in about the same region. For nitrate, the patterns of deposition are similar (figure 3), but the rate of deposition is about one-half the rate of sulfur deposition. Broad regions of the eastern U.S. receive deposition of nitrates in excess of 20 meq/m^2-yr, with peak deposition in excess of 40 meq/m^2-yr. Of the total sulfate and nitrate deposited in the eastern U.S., about two-thirds is sulfate.

By comparing figures 1 through 3, one can see that the deposition of acid-producing substances (sulfate and nitrate) is about 25-50% greater than the deposition of acidity. The remainder of the sulfate and nitrate are neutralized by other constituents, such as calcium and ammonium.

If significant reductions are to be made in the deposition of acidity, reductions in sulfate deposition might be the most likely way to begin, given that about twice as much sulfate is present in rainfall as nitrate. Similar patterns are assumed to occur for dry deposition of acid-producing substances, but far fewer observations are available to substantiate this.

As mentioned previously, precipitation is one of the sources of acid-producing substances, but scientists are less concerned about the acidity of rainfall than about the amount of acid-producing substances that eventually

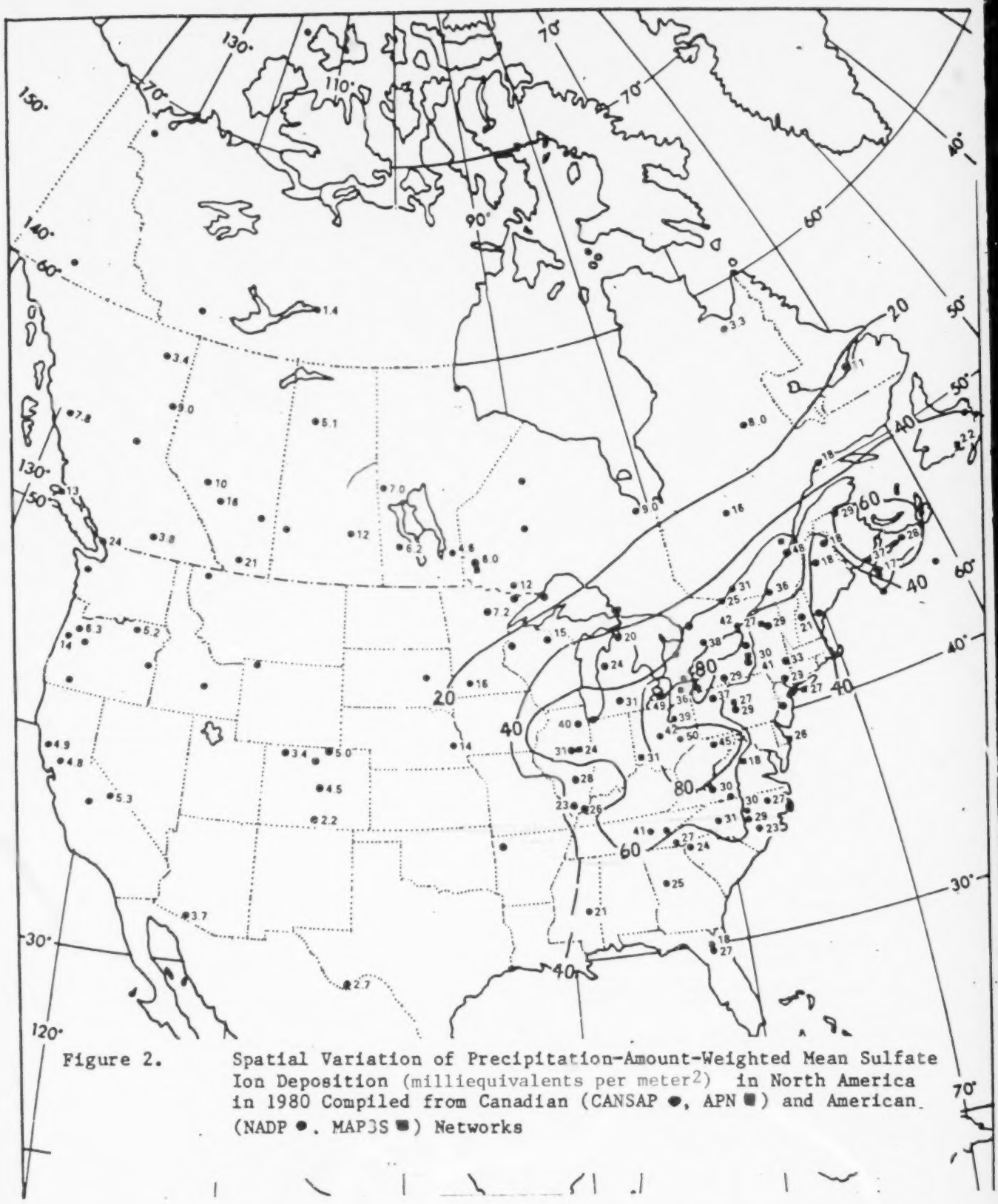


Figure 2. Spatial Variation of Precipitation-Amount-Weighted Mean Sulfate Ion Deposition (milliequivalents per meter^2) in North America in 1980 Compiled from Canadian (CANSAP ●, APN ■) and American (NADP ●, MAP3S ■) Networks

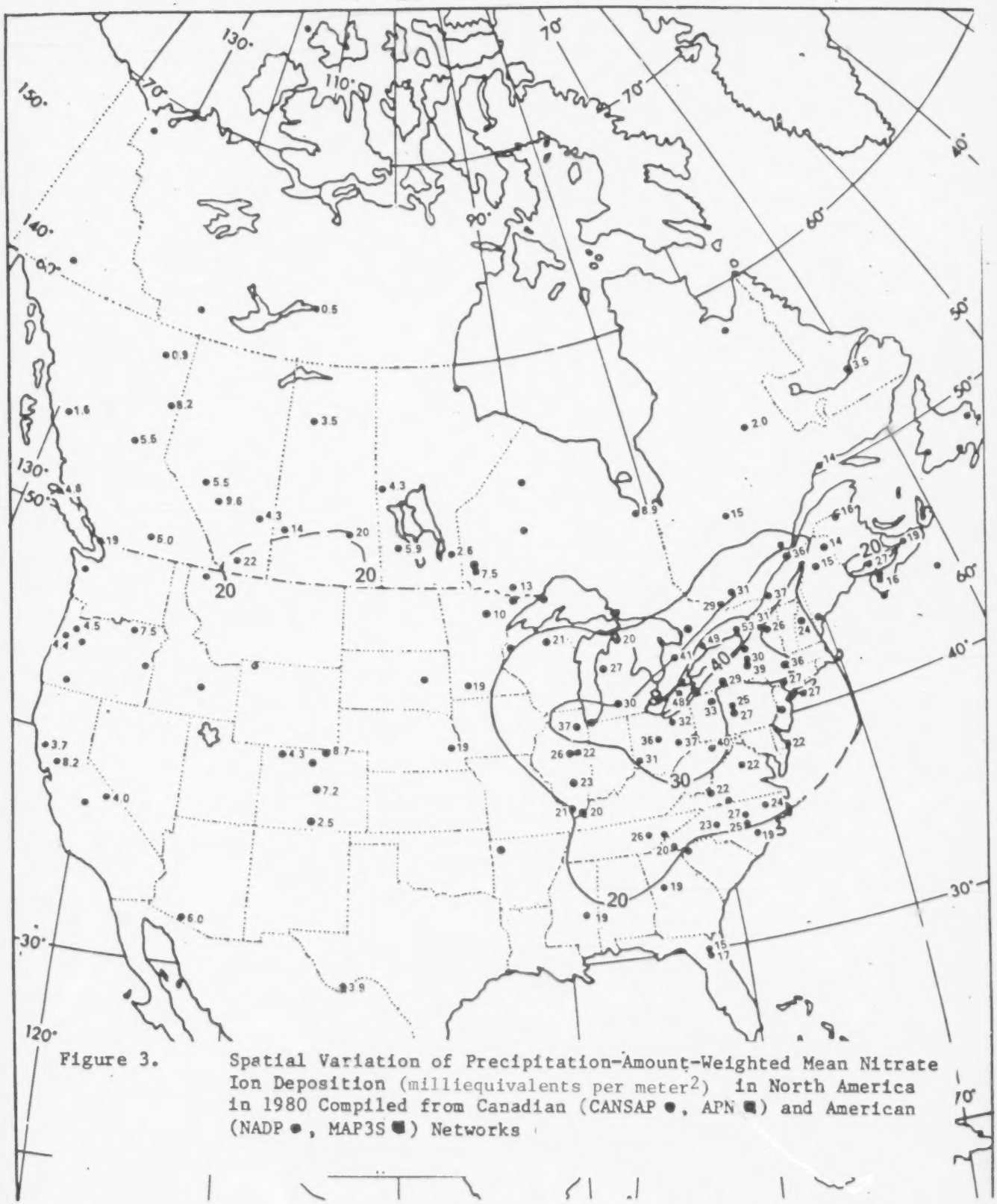


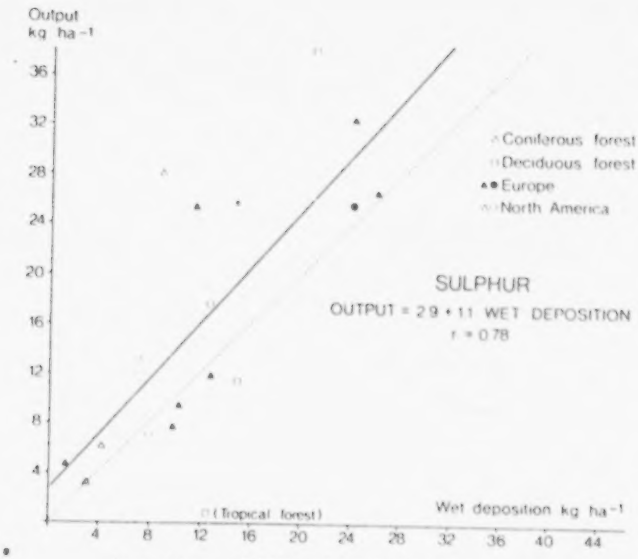
Figure 3. Spatial Variation of Precipitation-Amount-Weighted Mean Nitrate Ion Deposition (milliequivalents per meter^2) in North America in 1980 Compiled from Canadian (CANSAP ●, APN ■) and American (NADP ●, MAP3S ■) Networks

make their way to and through the environment. For potential impacts such as materials damage, the acidity of rainfall may be a useful indicator of environmental damage. For alterations of lake and stream water quality, the acidity of water after it flows through a surrounding watershed is of greater consequence.

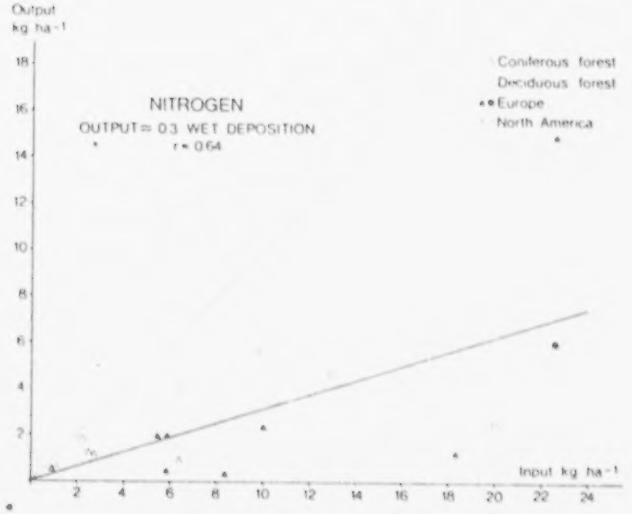
Figure 4 illustrates the relationship between the quantity of acid-producing substances in rainfall versus the quantity of acid-producing substances in water flowing out of a watershed. Figure 4A shows that the amount of sulfate leaving a watershed is about equal to, and in many cases greater than, the amount entering a watershed in rainfall, when averaged over a period of one or several years. That more sulfur leaves a watershed than enters from rainfall is probably due to the unaccounted amounts that enter from dry deposition of gases and particles.

For nitrogen-containing substances such as nitrate and ammonium (as shown in figure 4B), the picture is different. Only about 1/3 of the nitrogen entering a watershed from rainfall leaves the watershed. If dry deposition is considered, the sum of nitrogen input is greater, making the ratio of nitrogen leaving to nitrogen entering a watershed even lower. As discussed in the forest resources section, these results might be expected -- while both nitrogen and sulfur are essential nutrients for forest growth, the requirements for nitrogen are much greater than the need for sulfur in most eastern forests.

In summary, when considering the acid-producing substances in rainfall (and effects that are directly related to rainfall acidity), sulfate is generally present in quantities twice as great as nitrate. When considering the acid-producing substances that travel through the environment (and effects



A: *Annual input of SO_4^{2-} – S by precipitation versus the leaching losses from forest watersheds and lysimeters with forest soils. Solid line is the regression line, dotted line is the 1:1 line.*



B: *Annual input of $(NH_4^+ + NO_3^-)$ – N by precipitation versus the leaching losses from forest watersheds and lysimeters with forest soils. Solid line is the regression line, dotted line is the 1:1 line. Data from studies in Europe and North America*

Figure 4: Sulfur and nitrogen: quantities in precipitation versus outflow from watersheds (from: G. Abrahamsen, "Acid Precipitation, Plant Nutrients, and Forest Growth" (1980). Ecological Impact of Acid Precipitation, Proceedings of an International Conference.)

such as alterations of lake and stream water quality), sulfate may predominate even more. This can vary from region to region, and certainly does vary for short periods (for example, during the spring snow melt, the acidity from nitrate can be as great as or even greater than the acidity from sulfate in some watersheds), but over broad regions on an annual time scale, sulfates comprise a much larger share of acid-producing substances than nitrates.

III. MAN-MADE AND NATURAL SOURCE CONTRIBUTIONS TO ACIDIC DEPOSITION:

Averaged over large regions (for example the eastern U.S. or the North American continent), the background levels of acidic deposition due to natural sources (and possibly global-scale transported air pollutants) are thought to be relatively uniform. Some localized variation does of course occur, such as sulfate deposition from sea-spray along coastlines, but the relative importance of deposition from natural background sources versus deposition from regional man-made sources is more closely related to the magnitude of man-made pollutant emissions.

The natural background of wet sulfur deposition in North America (excluding Mexico) has been estimated to be about 4 to 10 meq/m^2*. This is about 20-40% of the average sulfur deposition from precipitation over all of North America. In areas of highest deposition the percentage is smaller. Averaged over eastern North America (east of the Mississippi and south of James Bay in Canada), natural background sources might contribute about 12-25% of the total. Over parts of the eastern U.S., where wet sulfur deposition exceeds 50 meq/m^2 and in areas is as high as 80 meq/m^2, the natural background

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* R. B. Husar and J. M. Holloway "Sulfur and Nitrogen Over North America". Presented at 1982 Stockholm Conference on Acidification of the Environment.

contributes even less.

This natural background originates from several different sources: Husar estimates that 6% of the wet sulfur deposition in North America originates from sea-spray. EPRI has estimated that about 5% of the sulfur dioxide emissions originate from natural, biological sources.* In addition, geologic sources such as volcanoes are thought to contribute to natural background.

Even averaged over the North American continent, the acid-producing potential of man-made sulfur dioxide emissions far exceeds total wet sulfur deposition. The over 30 million tons of sulfur dioxide emitted from man-made sources in North America is 2.5 times the total wet sulfur deposition and 6 to 10 times the natural background. The remaining portion of man-made sulfur is either deposited dry or exported off the continent.

For wet nitrate deposition, the percentage of natural background to total deposition is estimated to be similar to sulfur. Natural background is estimated to contribute about 20-40% of wet nitrate deposition when averaged over the North American continent, and about 10 to 25% when averaged over eastern North America.

IV. THE RELATIONSHIP BETWEEN CURRENT EMISSIONS AND DEPOSITION:

One of the major controversies in the acid rain debate is the effect that emissions of pollutants from any source (or group of sources) will have on ambient air quality and pollutant deposition at some other location. The relationship between emissions and deposition is determined by a complex chain

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* EPRI, "Biogenic Sulfur Emissions in the SURE Region" EA-1516, Electric Power Research Institute (1980).

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of chemical and physical processes which occur as primary pollutants (for example, sulfur dioxide) are emitted, transformed into secondary pollutants (e.g., sulfates), transported, and finally deposited.

Computer models, called transport models, are used to simulate the sequence from emissions to transformation, transport, and deposition mathematically. Transport models are the only practical procedure available to estimate the relationship between areas of origin and areas of deposition of long-range transport pollutants. Large-scale regional transport cannot now be measured directly for the large number of sources of emissions and deposition regions of interest, and under the variety of meteorological conditions needed to perform the analysis. Models describing long-range transport of sulfur oxides have been available for several years; preliminary models of nitrogen oxides transport are just now being developed.

The major long-range transport (LRT) models describing acid deposition incorporate six atmospheric processes: release of emissions, horizontal transport and dispersion, vertical mixing of pollutants in the atmosphere, chemical transformation, dry deposition, and wet deposition. The models themselves are composed of sub-models which simulate chemical and physical processes within these broader categories. The main data requirements of LRT models are: emissions inventories of pollutants, meteorological data, ground cover data, and a host of values (parameters) representing chemical reaction rates and other physical processes.

During the spring of 1981, OTA requested assistance from the Atmospheric Modeling Committee of the Utility Air Regulatory Group (UARG) for review of several LRT models. The review, performed by a contractor for UARG, assessed the representation of each of the major processes listed

above for five major LRT models.* The review pointed out that: 1) The models consider only sulfur compounds and not the other constituents of acid deposition; 2) The models reviewed vary widely in their analytical treatment of critical processes; 3) Model evaluation is in a preliminary stage and the existing data bases do not allow critical evaluation of important features of the models; and 4) The merits of the methodology must remain in question until the effects of certain complex chemical reactions (such as the effects of nitrogen oxides on sulfur oxide reactions) are included in the model.

Although the UARG review concluded that due to the reasons listed above, the "models cannot, at the present time, provide adequate information that would assist in distinguishing between policy options", the review also states that "the best available methodology currently available for investigating the transport, transformation, and deposition of atmospheric pollutants on a regional scale involves the use of long-range transport models."

Modeling results are presented in this section, therefore, to provide a plausible description of the current relationship between sources and deposition of sulfur oxides in the eastern U.S. As the UARG review points out, this is only one of the acid-producing substances currently deposited; however, as discussed earlier, sulfate is currently the major acid-producing substance in precipitation.

The model used in this analysis is the Advanced Statistical Trajectory Regional Air Pollution Model (ASTRAP) developed by Argonne National Laboratory under DOE and EPA funding. The UARG review rates the ASTRAP model as the most

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* C. Shepard Burton et.al., "A Review of Certain Long-Range Atmospheric Transport Models and the Source-to-Sink Transfer Matrix Methods", Systems Applications, Inc., May 1981.

sophisticated of the models considered. The ASTRAP model includes several components of the source-receptor relationship felt to be important by the UARG review but not included in the other models.*

Since the UARG review, some progress has been made in evaluating the ASTRAP model. Results of the evaluation activities sponsored under the U.S.-Canada Memorandum of Intent (MOI) on Transboundary Air Pollution are expected to be released shortly. Preliminary results, presented at a recent scientific symposium,** indicate that the eight models evaluated (including ASTRAP) appear to reproduce the large-scale patterns of observed wet deposition. As pointed out in the UARG review, a complete evaluation of LRT models cannot be performed at this time because of the size and quality of the data base.***

The last point raised in the Utility Air Regulatory Group review -- that the effects of complex non-linear chemistry and co-pollutants such as nitrogen

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* The ASTRAP model includes: the ability to account for seasonal differences (the ASTRAP model simulates January and July average conditions); release height of emissions; the use of detailed meteorological data; consideration of vertical atmospheric processes in addition to horizontal movement; wet deposition rates that vary with rainfall intensity; and dry deposition rates that vary temporally and spatially. The other models evaluated include some of these components, but ASTRAP is the only model that includes all of them.

** H. L. Ferguson and J. W. S. Young "Atmospheric Long Range Transport Models: Science and Policy Implications". Invited paper presented at AGU Symposium on Acid Rain, June 1982.

*** Even if the models were "perfect", one would expect the model simulations to deviate from the observations since the latter are influenced by both the factors treated in the model and the factors that are not treated (e.g., small-scale precipitation variations). In addition, part of the difference between model simulations and obvservations is due to the inherent variability of the real world. The model is of necessity designed to simulate an average of the observations while the monitoring data base at this time is insufficient to calculate a representative average (i.e., the average over a number of years with similar emissions, meteorological conditions, etc.).

oxides are not included in ASTRAP -- will be discussed in greater detail in the next section. OTA has used another model that incorporates more realistic (non-linear) atmospheric chemistry and considers the effects of co-pollutants such as nitrogen oxides and hydrocarbons (but at the expense of sophisticated meteorology), to gain preliminary insights about the simplifying chemical assumptions used in ASTRAP. The results of this comparison indicate that the ASTRAP model might adequately represent dry deposition of sulfur, but that wet deposition of sulfur might be significantly affected by variations in ambient concentrations of other pollutants. (These results will be discussed in the following subsection.)

Because of its sophisticated treatment of the regional patterns of emissions and meteorology, the ASTRAP model can be used to investigate the current relationship between regions of sulfur dioxide emissions and regions of sulfur deposition. The model is best used in a relative sense -- for example, estimating the percent contribution from one region to deposition in another -- rather than for making quantitative projections of the magnitude of deposition. Again the discussion of sulfur deposition that follows must be considered as a description of current relationships, interpreted as general patterns rather than quantitative predictions.

Acid deposition has often been characterized as "acid in Adirondack lakes from power plants in Ohio." The ASTRAP model can be used to show that such statements are overly simplistic: pollutants do travel from one region to another, but in all directions, not just west to east. In addition, while pollutants can travel long distances, emissions within a region contribute a large share to total deposition in that region.

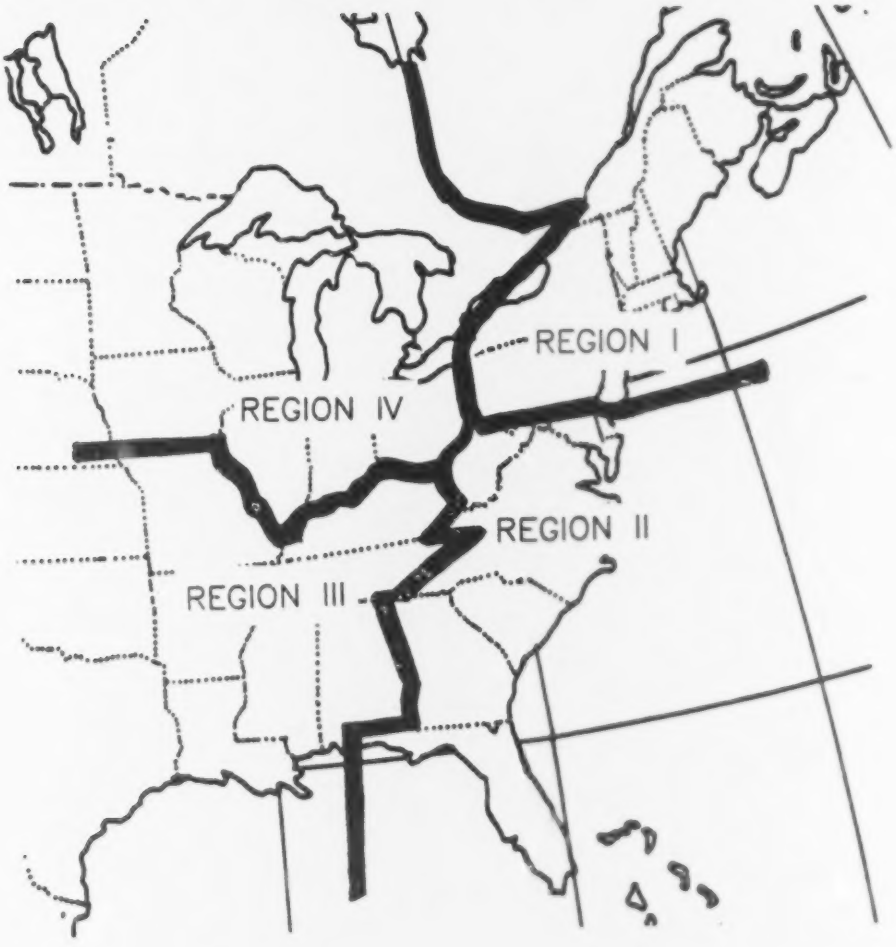
To illustrate these points, eastern North America can be divided into

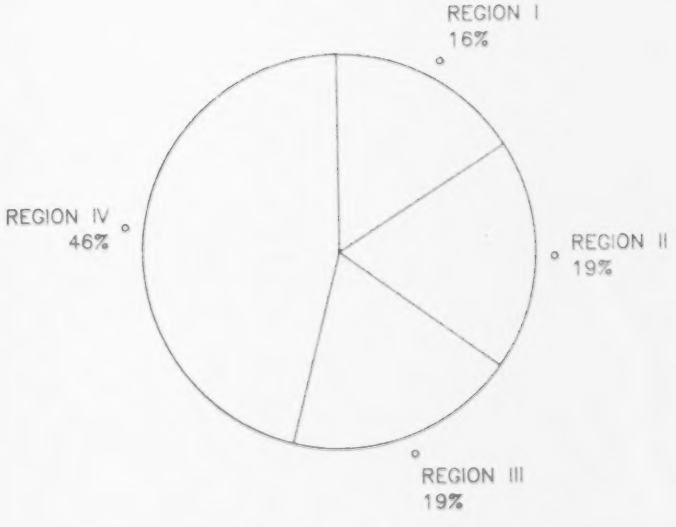
four regions as shown in Figure 5. The intersection of the regions has been chosen to correspond to the area of peak wet sulfur deposition in 1980 so as to not overly bias deposition into any one region. Figure 5 also displays the percentage of sulfur dioxide emitted in each region, and the percentage of total sulfur deposited (as simulated by ASTRAP) in each of the regions. Sulfur dioxide emissions are roughly comparable in the northeastern region (I), southeastern region (II), and southwestern region (III). Emissions in the northwestern region (IV) are over twice the amount of any of the other regions. Deposition is lowest in the southern regions (II and III) and highest in the northern regions (I and IV).

Figures 6 through 9 show model results for: 1) the percentage of deposition in each region originating from within its borders, and from each of the other regions, and 2) of the deposition originating outside a region, the percentage of deposition originating within 500 km, 500-1000 km, 1000-1500 km, and from distances greater than 1500 km from the area of deposition.

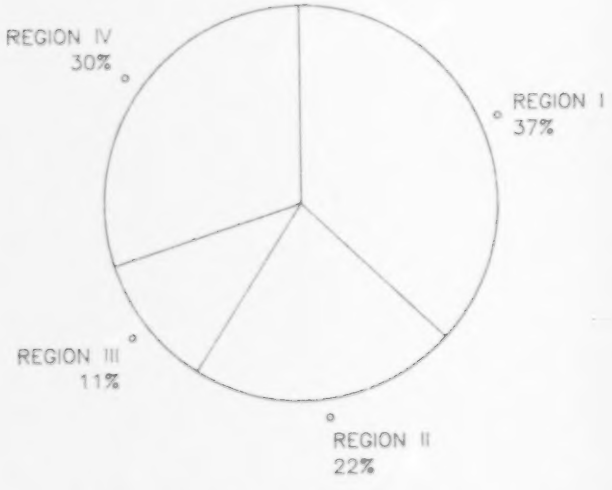
For example, figure 6 illustrates these relationships in region I (the northeastern region). The pie chart in the upper right illustrates the model projection that approximately 80% of the deposition comes from emissions in two regions -- from within its own borders and from the northwestern region in about equal amounts. The bar graphs placed in regions II, III, and IV illustrate the percentage of deposition contributed from each region to the northeastern region, according to its distance from the sources of emissions. For example, in the lower right, the bar graph shows model estimates that 40% of the deposition from the southeastern region originates within 500 km of the eventual area of deposition; another 40% originates between 500 to 1000 km away, and the remainder travels over 1000 km from

Figure 5





SULFUR DIOXIDE EMISSIONS
PERCENT BY REGION



TOTAL SULFUR DEPOSITION
PERCENT BY REGION

Figure 6: Source-Receptor Relationships Calculated by the ASTRAP Model

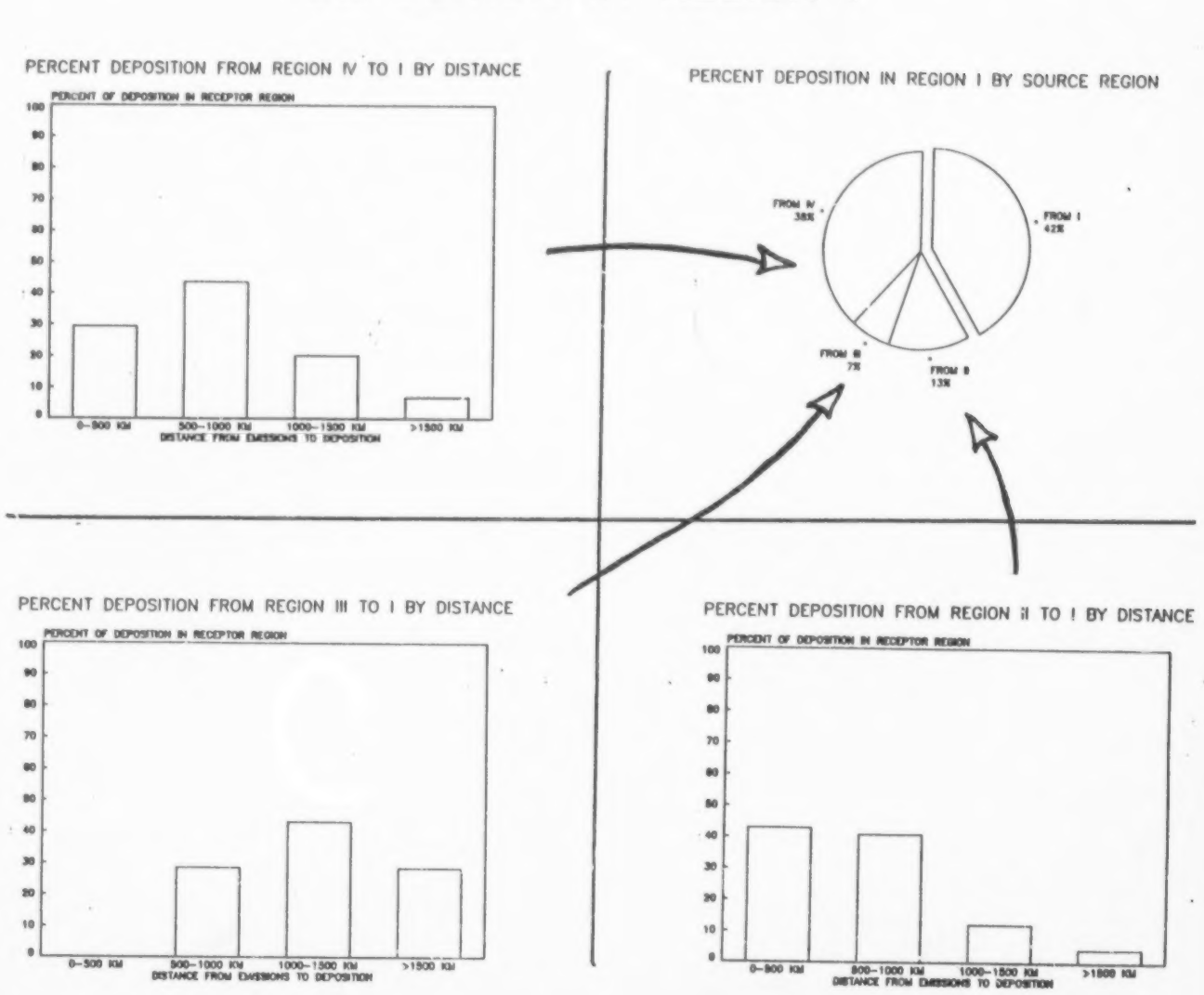

DEPOSITION IN REGION I
PERCENT DEPOSITION FROM REGION IV TO I BY DISTANCE
PERCENT OF DEPOSITION IN RECEPTOR REGION
0-900 KM
500-1000 KM
1000-1500 KM
>1500 KM
DISTANCE FROM EMISSIONS TO DEPOSITION
PERCENT DEPOSITION IN REGION I BY SOURCE REGION
FROM IV 38%
FROM I 42%
FROM III 7%
FROM II 13%
PERCENT DEPOSITION FROM REGION III TO I BY DISTANCE
PERCENT OF DEPOSITION IN RECEPTOR REGION
0-500 KM
900-1000 KM
1000-1500 KM
>1500 KM
DISTANCE FROM EMISSIONS TO DEPOSITION
PERCENT DEPOSITION FROM REGION II TO I BY DISTANCE
PERCENT OF DEPOSITION IN RECEPTOR REGION
0-900 KM
900-1000 KM
1000-1500 KM
>1500 KM
DISTANCE FROM EMISSIONS TO DEPOSITION

Figure 7: Source-Receptor Relationships Calculated by the ASTRAP Model

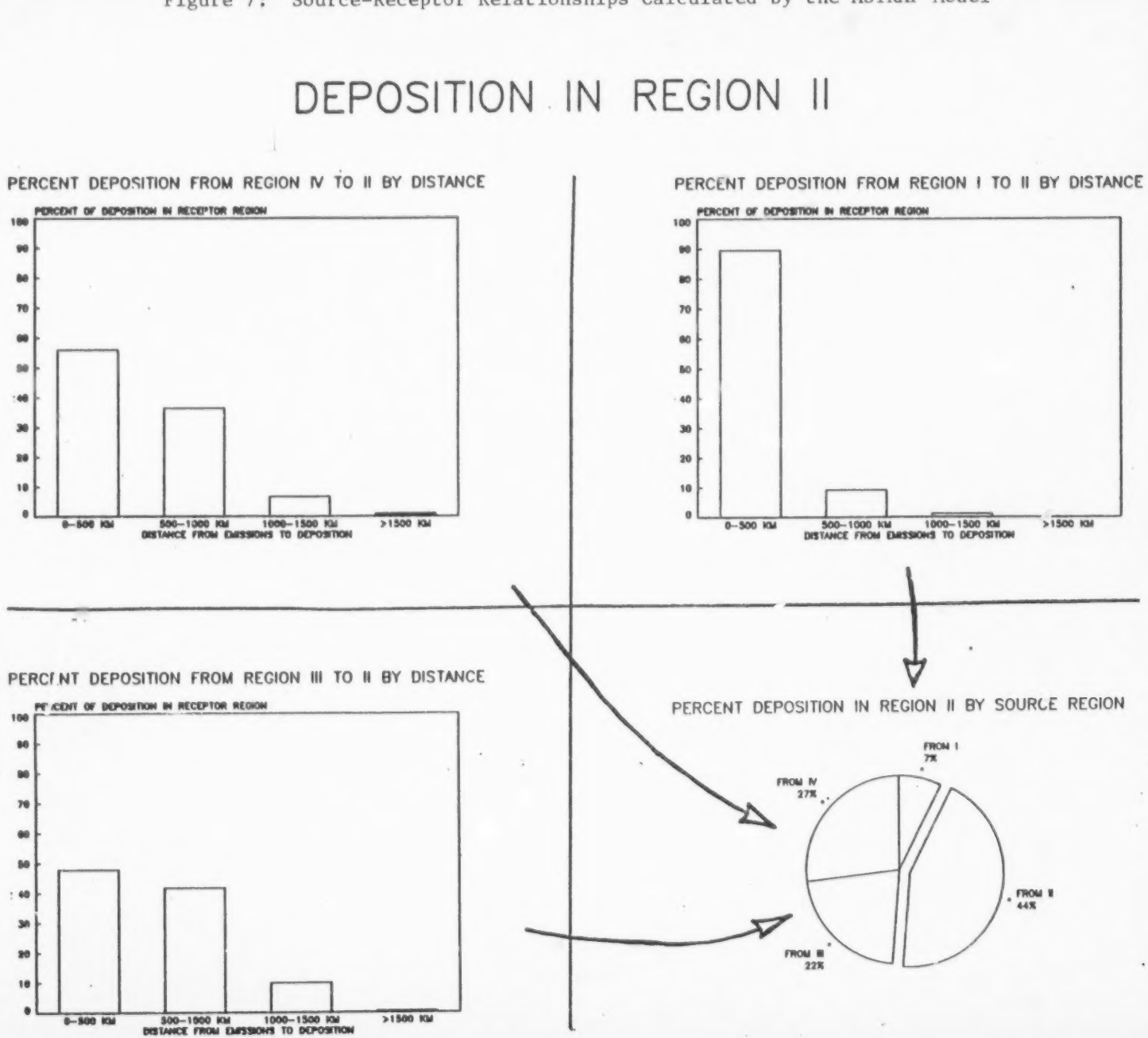


Figure 8: Source-Receptor Relationships Calculated by the ASTRAP Model

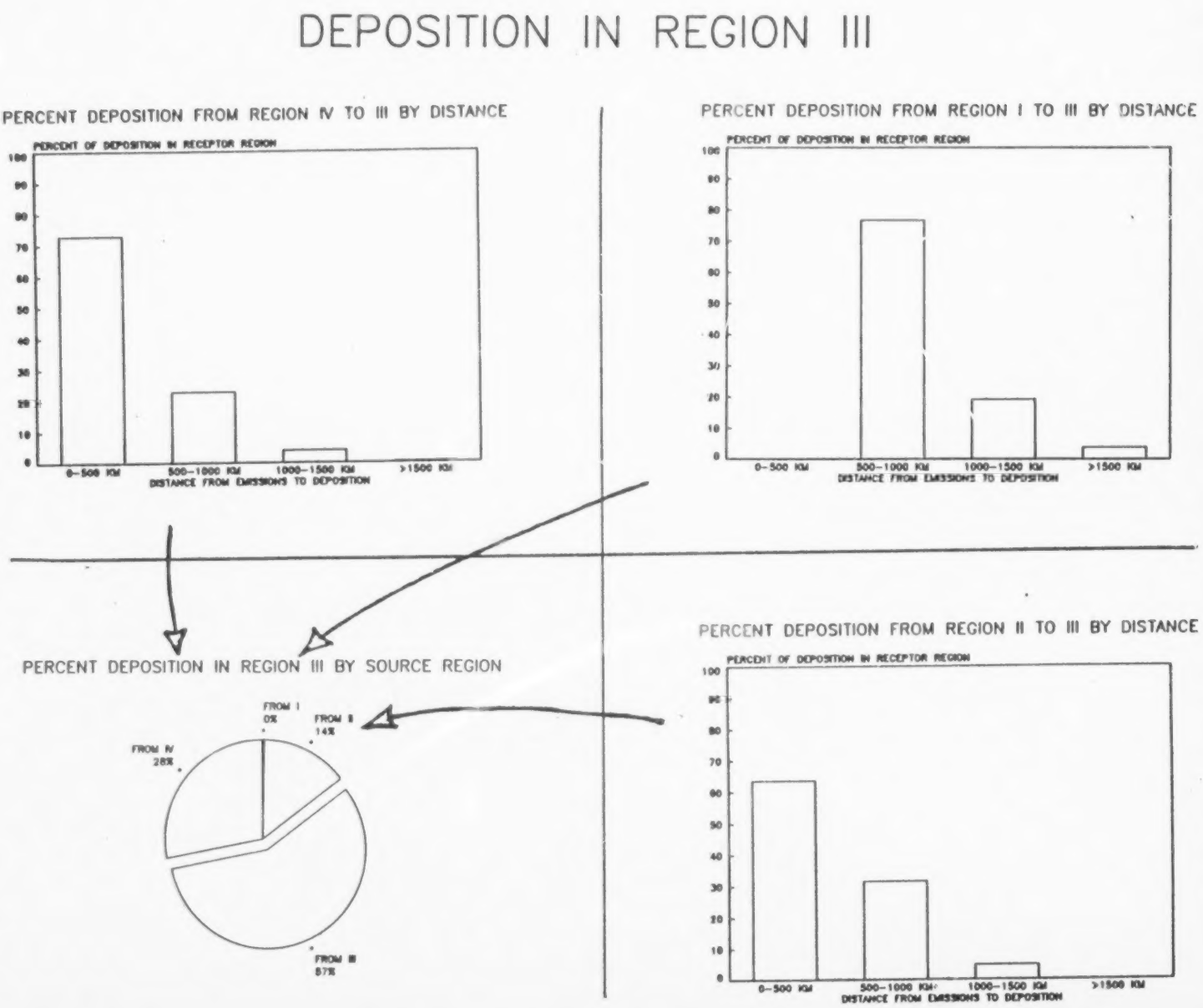
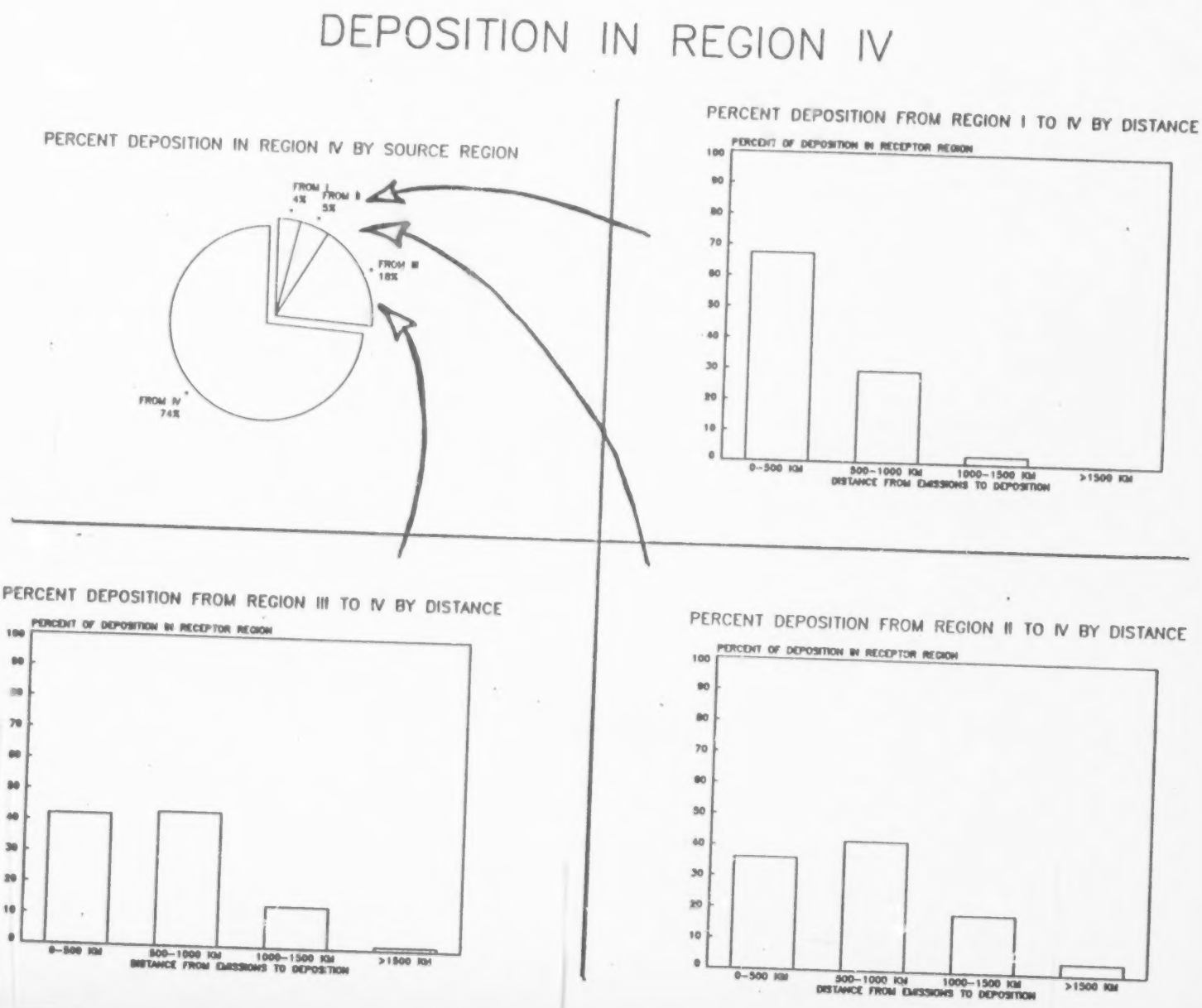


Figure 9: Source-Receptor Relationships Calculated by the ASTRAP Model


DEPOSITION IN REGION IV
PERCENT DEPOSITION IN REGION IV BY SOURCE REGION
FROM I 4%
FROM II 5%
FROM III 16%
FROM IV 74%
PERCENT DEPOSITION FROM REGION I TO IV BY DISTANCE
PERCENT OF DEPOSITION IN RECEPTOR REGION
0-500 KM
500-1000 KM
1000-1500 KM
>1500 KM
DISTANCE FROM EMISSIONS TO DEPOSITION
PERCENT DEPOSITION FROM REGION III TO IV BY DISTANCE
PERCENT OF DEPOSITION IN RECEPTOR REGION
0-500 KM
500-1000 KM
1000-1500 KM
>1500 KM
DISTANCE FROM EMISSIONS TO DEPOSITION
PERCENT DEPOSITION FROM REGION II TO IV BY DISTANCE
PERCENT OF DEPOSITION IN RECEPTOR REGION
0-500 KM
500-1000 KM
1000-1500 KM
>1500 KM
DISTANCE FROM EMISSIONS TO DEPOSITION

emissions to deposition.

By comparing figures 6 through 9, some general observations can be made:

1. All regions contribute to deposition in all other regions.

2. At the spatial scale used in this analysis, each region contributes as much or more to its own deposition as any other single region contributes to it.

3. Substantial quantities of deposition originate from emissions over 500 km away.

4. Pollutants are transported longer from west to east and south to north than from east to west or north to south.

V. THE EFFECTIVENESS OF EMISSIONS REDUCTIONS FOR ACHIEVING DEPOSITION REDUCTIONS

Inherent in the use of regional-scale models such as ASTRAP are assumptions concerning the physical and chemical processes involved in the transformation of sulfur dioxide to sulfate. Linear regional-scale models assume that the production of sulfate is proportional to the concentration of sulfur dioxide. The source-receptor relationships produced by these models can be used to describe expected changes in deposition only to the extent that the changes in deposition are linearly proportional to changes in emissions.

While the source-receptor relationships shown before may represent reasonable estimates of inter- and intra-region transport, the use of these results for control strategies may be misleading unless the chemical transformation and deposition system can be shown to behave in a linear manner. This section first discusses the results of linear model runs for different emissions scenarios and then presents an initial evaluation of how the addition of more realistic atmospheric chemistry might alter the results.

OTA asked the Argonne National Laboratory to use the ASTRAP model to

simulate atmospheric concentrations and surface deposition of sulfur pollutants for three sulfur dioxide emissions scenarios. These include: current emissions of sulfur dioxide in the U.S. and Canada, emissions as allowed by S. 1706 (the Mitchell bill -- a 10 million ton reduction of sulfur dioxide in the eastern U.S.), and emissions as allowed by S. 1709 (the Moynihan bill -- an 8 million ton reduction of sulfur dioxide).* These scenarios are based on the assumption that reductions in emissions from each point source within a state are proportional to the reduction assigned that state in the OTA analysis** of the Mitchell and Moynihan bills.

The simulations were performed for 2 months, January and July, to investigate winter versus summer conditions. Figure 10 displays model projections of percentage reductions in both wet and dry sulfur deposition due to a 10 million ton reduction of sulfur dioxide emissions (S. 1706). The reductions shown apply only to emissions originating in the U.S. (i.e., deposition from emissions in Canada are not considered). Similar projections for an 8 million ton reduction (S. 1709) are shown in Figure 11. The pattern of deposition reductions is similar for both reduction formulae. The largest deposition reductions occur in the Midwest -- the region with both 1) the highest current emissions and 2) the greatest percentage of required reductions under both bills. The percentage reductions in deposition decline with distance away from this region. Because of the linear relationship between emissions and deposition assumed in the ASTRAP model, both scenarios show a percentage reduction in total deposition almost equal to the percentage

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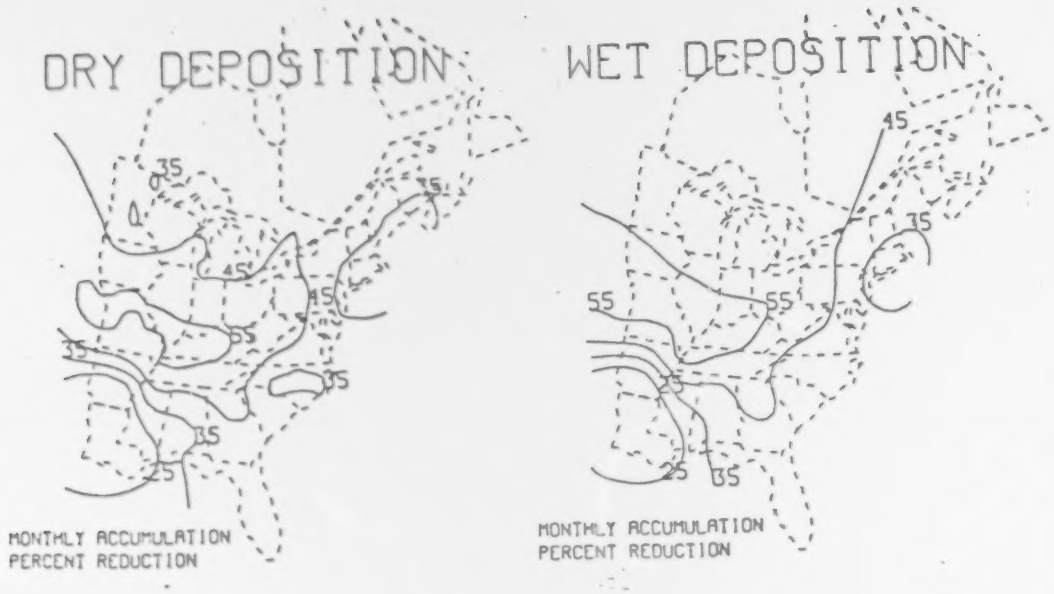
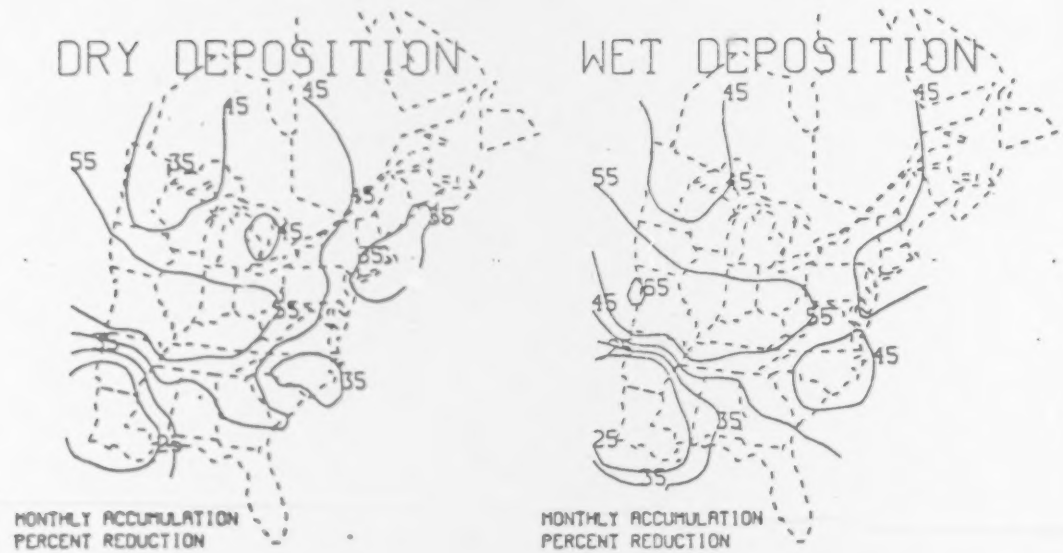
* Personal communication, J. Shannon, Argonne National Laboratory, December 1981.

** OTA estimates of state-by-state emissions allowed under S. 1706 and S. 1709 are presented in another section of this report.

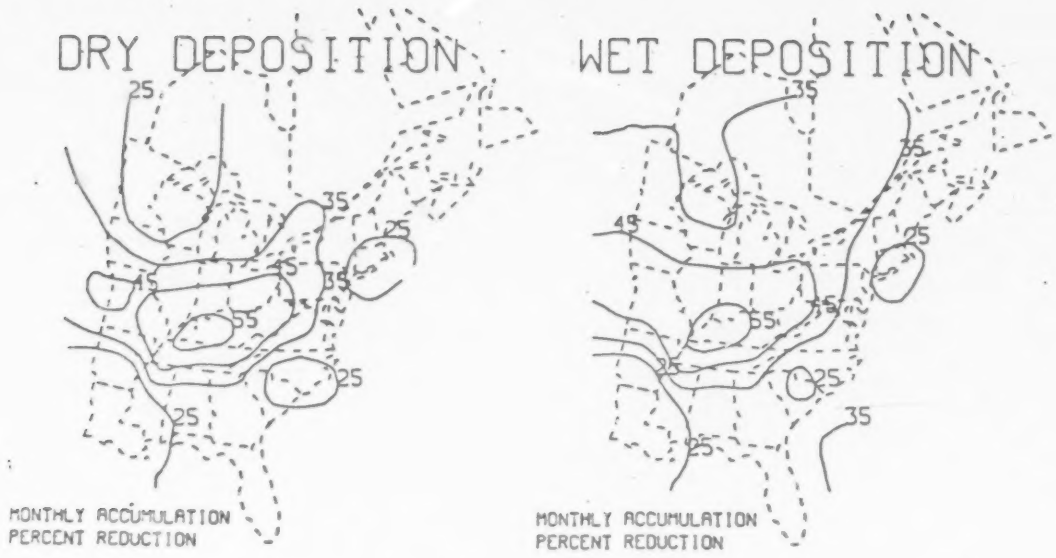


MITCHELL BILL, JANUARY

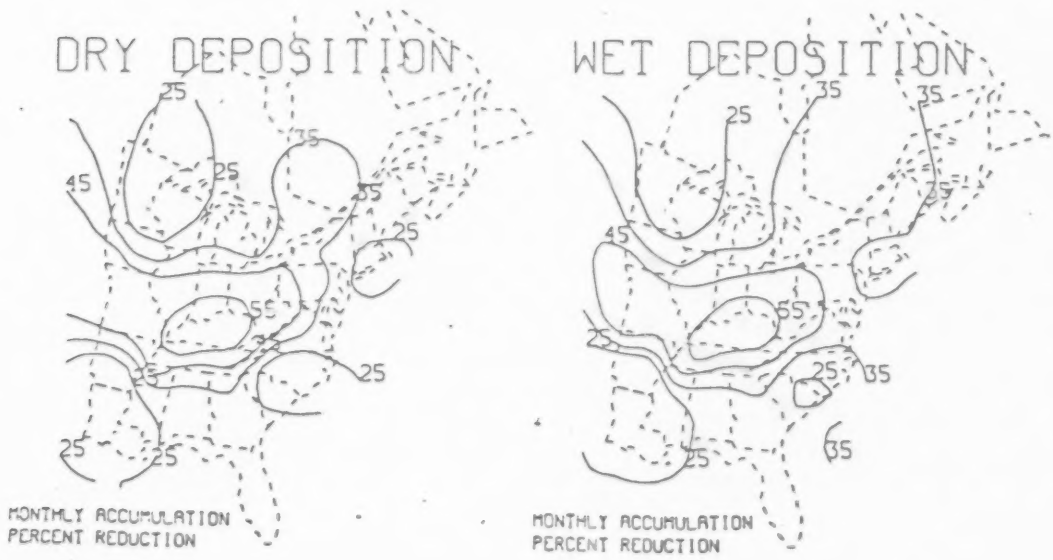


MITCHELL BILL, JULY

Figure 10: Percent reduction in sulfur deposition from 10 million tons sulfur dioxide emissions reduction allocated by S. 1706. (ASTRAP model results using OTA emissions analysis).



MOYNIHAN BILL, JANUARY



MOYNIHAN BILL, JULY

Figure 11: Percent reduction in sulfur deposition from 8 million tons sulfur dioxide emissions reduction allocated by S. 1709. (ASTRAP model results using OTA emissions analysis.)

reduction in total emissions when averaged over the entire eastern U.S.

The assumed linear relationship -- that is, a specified percentage reduction in emissions will lead to the same percentage reduction in deposition -- is a simplification adopted for computational advantages. No one would argue that the simplified relationship used in the model is a realistic representation of the complex transformations occurring in the atmosphere. Over 100 chemical reactions may potentially play a role in the transformation of sulfur dioxide to sulfate. Unfortunately, a complete evaluation of the chemical transformation process is not available, since the rate constants for many of the equations are not known. The importance of understanding the full ensemble of chemical equations cannot be currently evaluated. A more pertinent question is: how accurate is the approximation of a linear relationship over the large time and space scales involved in region-wide emissions and deposition?

Various authors have speculated on the sensitivity of deposition to changes in emissions in the eastern United States. Clearly the total amount of sulfur emitted to the atmosphere is eventually deposited; thus, ultimately, reductions in deposition will approximate the reduction of emissions. (Deposition from natural sulfur sources, though small, would remain.) The crucial question is where the reductions in sulfur deposition would occur -- in about the same regions as predicted by the linear models, closer to emission sources, or further away.

To examine the linearity of the chemical transformation system, a second model, developed by Rodhe et al. (1981), was run for OTA by a contractor at the University of Michigan.* The Rodhe model has the capability to simulate the interactions of several pollutants (rather than just sulfur dioxide as in

the ASTRAP model) that may affect the deposition of acidic substances.

This technique was chosen because it allows a reasonable qualitative evaluation of pollutant interactions while remaining computationally tractable. The model is limited in its description of atmospheric mechanisms (such as mixing of pollutants with surrounding air), which could affect the quantitative results it predicts. In addition, it does not incorporate the meteorology necessary to describe complex patterns of deposition. Nonetheless, this system represents a useful step forward from linear modeling, as it has the capability of allowing for chemical transformations based on a series of 19 chemical equations. The use of the Rodhe model yields a qualitative picture of the potential influence of changes in other primary pollutants -- principally reactive hydrocarbons (RHC) and nitrogen oxides (NO_x) -- on downwind deposition.

Dry sulfur deposition is highly dependent on the concentration of sulfur dioxide in the atmosphere. Changes in RHC and NO_x might alter the conversion of sulfur dioxide to sulfate and change the reservoir of SO_2 available for dry deposition. However, the model simulations indicate that dry sulfur deposition is fairly insensitive to the concurrent mix of NO_x and RHC in the atmosphere. Figure 12 illustrates this by showing the change in dry sulfur deposition per unit change of primary pollutant concentrations as a function of time downwind of a source region as predicted by the Rodhe model. The graph illustrates that, of the three pollutants considered, sulfur dioxide has by far the greatest effect on dry sulfur deposition. In addition, the figure shows that, for pollutant travel times of less than 2 days, the assumption

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* Perry J. Samson, "On the Linearity of Sulfur Dioxide to Sulfate Conversion in Regional-Scale Models". OTA contractor report, June, 1982.

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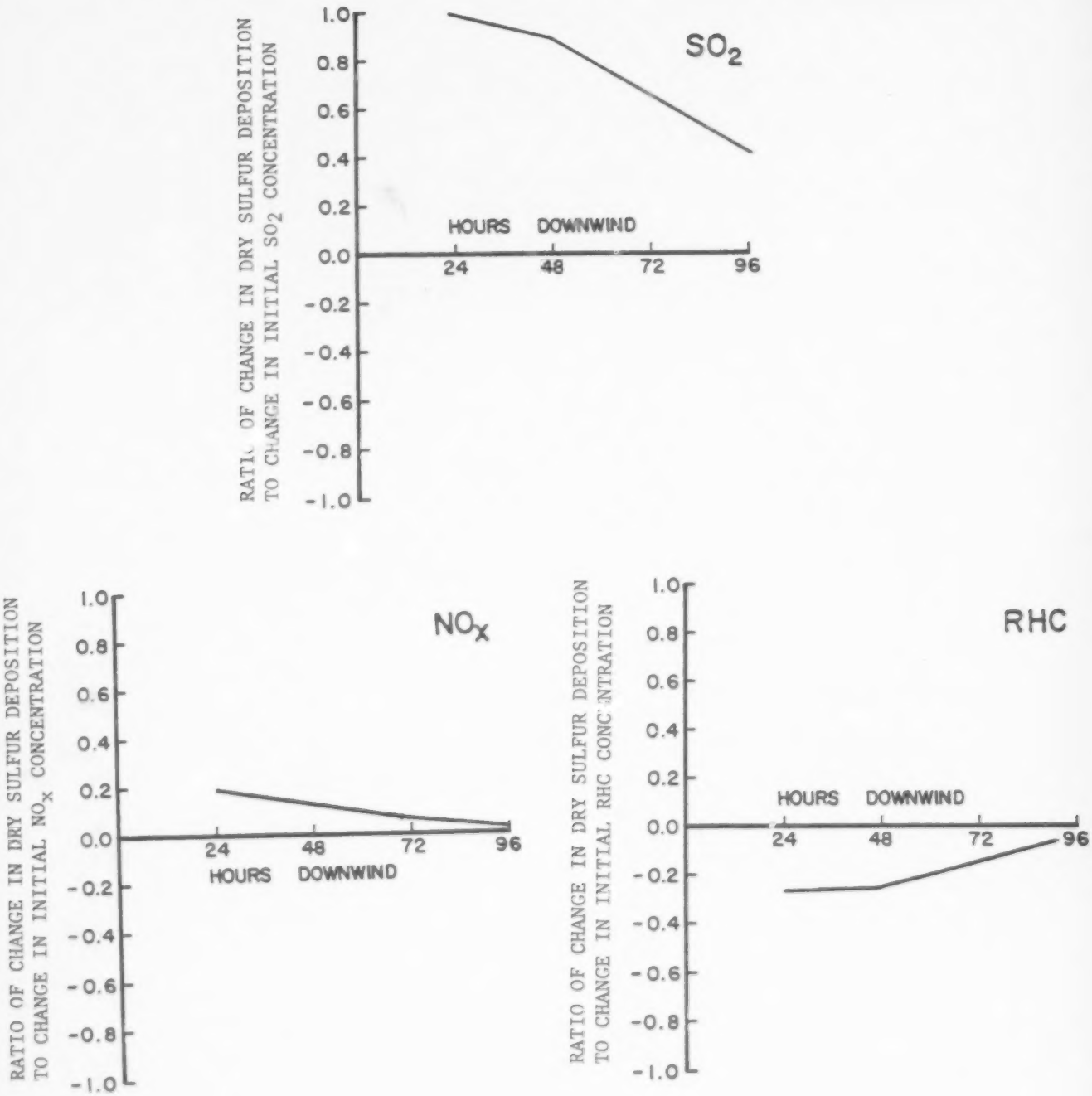

Figure 12: Ratio of the change in dry sulfur deposition to changes in initial concentrations of sulfur dioxide, nitrogen oxides, and reactive hydrocarbons, as a function of time downwind of the source region.

that a given percentage reduction in emissions will lead to a comparable reduction in dry sulfur deposition seems reasonable. Thus, because most of the dry deposition occurs within this time period, the omission of co-pollutants and the linear relationship assumed in regional-scale models such as ASTRAP may be satisfactory for predicting changes in the dry sulfur component of deposition.

The sensitivity of wet sulfur deposition to pollutant changes is considerably different than that of dry deposition. Its response, shown in Figure 13, is such that a 50 percent change in the initial amounts of sulfur dioxide will result in roughly a 30 percent change in wet sulfur deposition.* One must keep in mind that these results are highly dependent on the way chemical transformations in clouds are approximated by the model. Because such processes cannot yet be simulated in detail, the accuracy of this result cannot be evaluated at this time.

Similar to dry deposition, the wet deposition of sulfur is affected more by changes in sulfur dioxide concentration than by changes in other pollutant concentrations. It should be noted, however, that the sensitivity of wet sulfur deposition to changes in RHC is relatively high. Reductions in RHC could lead to reductions in wet sulfur deposition, but the effect would not be as pronounced as equivalent reductions in sulfur dioxide. The sensitivity of wet sulfur deposition to changes in NO_x concentrations is less pronounced, but in the opposite direction, i.e., reductions in NO_x might lead to a small increase in wet sulfur deposition. However, this increase might be offset by

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* This estimate is for: 1) relatively high initial concentrations of pollutants, such as those found in the Ohio River Basin during pollution episodes and 2) an "average" rainfall event (1 hour at 1 mm/hr) 24 hours downwind of the source.

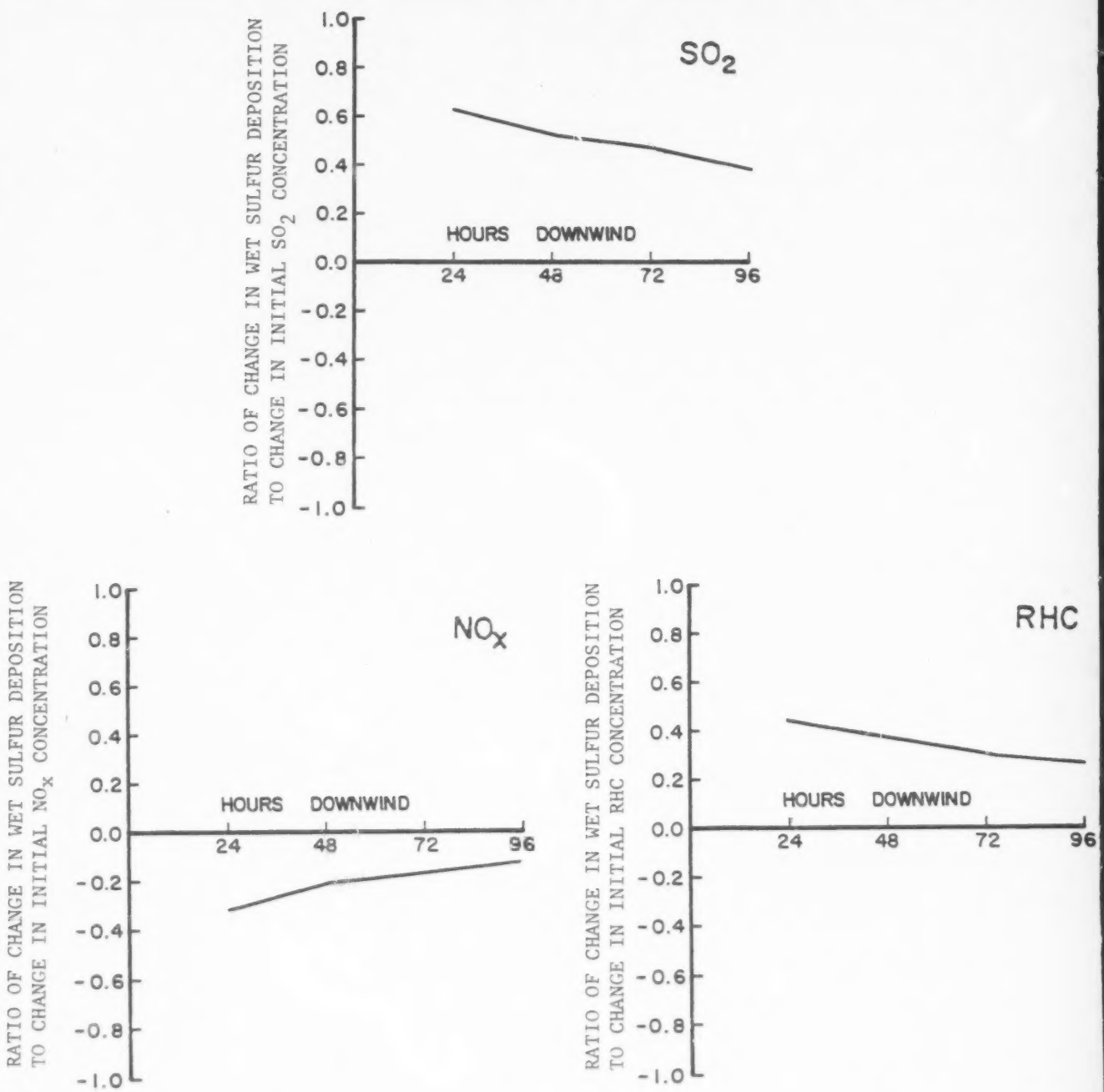

Figure 13: Ratio of the change in wet sulfur deposition to changes in initial concentrations of sulfur dioxide, nitrogen oxides, and reactive hydrocarbons, as a function of time downwind of the source region. (Assuming a precipitation rate of 1 mm/hr at a given time.)

reductions in dry deposition of sulfur and reductions in nitrate deposition.

The Rodhe model is not meant to be used for quantitative estimates of pollutant deposition. It was employed as a tool to evaluate the qualitative nature of pollutant interactions and how they might affect dry and wet sulfur deposition. The results sugggest that wet sulfur deposition is relatively sensitive to the mix of reactive hydrocarbons and nitrogen oxides while dry sulfur deposition is not.

The analysis suggests that if reducing total sulfur deposition is a desired goal, a control strategy might be effectively designed by reducing sulfur dioxide emissions. However, for reductions in wet sulfur deposition, concurrent changes in concentrations of nitrogen oxides and reactive hydrocarbons are also of concern.

VI. "TARGET" DEPOSITION RATES TO PROTECT SENSITIVE RESOURCES

The previous discussion was limited to considering the percentage reduction in total sulfur deposition that might occur from sulfur dioxide emissions reductions of 8 and 10 million tons per year in the 31-state eastern region. No attempt was made to relate these estimates to prevention of potential damage to sensitive resources. As of this writing three separate groups have estimated deposition "targets," i.e., levels of deposition below which acidification of sensitive resources is not expected to occur.*

The first of these targets was established by the Impact Assessment

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* OTA estimates of the effects of changes in deposition on acid-altered aquatic resources are presented in section G of this report. Rather than presenting specific deposition targets to prevent acidification of sensitive resources, the effects of three scenarios of changes in wet sulfur deposition are discussed.

Working Group established under the U.S.-Canada Memorandum of Intent on Transboundary Air Pollution. The Phase II Summary Report (October 1981) states:

> "Wet sulfate loadings values of 11-20 kg/ha/yr have been suggested as being protective for the majority of surface waters. At this loading, some of the most sensitive waters, i.e., those with low alkalinity, may still experience some water quality damage, particularly during snowmelt and when additional acids [nitric acid] are present."

In the recently published National Research Council/National Academy of Sciences report entitled "Atmosphere-Biosphere Interactions," the acidity of precipitation, rather than the concentration of sulfates, was used to specify a level that would protect sensitive freshwater ecosystems. NAS states:

> "It is desirable to have precipitation with pH values no lower than 4.6 to 4.7 throughout such areas, the value at which rates of degradation are detectable by current survey methods. In the most seriously affected areas (average precipitation pH of 4.1 to 4.2), this would mean a reduction of 50 percent in deposited hydrogen ions."

A third estimate was presented by Evans, Hendrey, Stensland, Johnson and Francis in a recent paper.* This group states:

> "For aquatic ecosystems, current research indicates that establishing a maximum permissible value for the volume weighted annual H^+ concentration of precipitation at 25 ueg/l may protect the most sensitive areas from permanent lake acidification. Such a standard would probably protect other systems as well."

This last estimate is quite similar to the National Academy estimate, as a H^+ concentration of 25 ueq/l is equal to a pH of 4.6.

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* L. S. Evans, et al. "Acidic Deposition: Considerations for an Air Quality Standard". Water, Air and Soil Pollution 16 (1981) 469-509.

To put each of these estimates in perspective, the deposition monitoring data presented in figures 1 to 3 can be used to derive maps of the percent reduction in wet sulfur deposition required to reach each of these targets. The three statements can be summarized into four plausible targets:

1) 20 kg/ha (42 meq/m^2) wet sulfate
2) average precipitation pH of 4.6 (25 ueq/l H^+)
3) average precipitation pH of 4.7
4) 50% reduction in H^+ (hydrogen ion) deposited in rainfall

It should be noted that, because of the larger data base from which to draw comparisons to ecological effects, these target values all use wet deposition as a surrogate for total (wet plus dry) deposition of acidifying substances.

To calculate the reduction in sulfate deposition to reach the last three target values, several assumptions must be made. First, one must assume that if wet sulfate levels are reduced, a corresponding quantity of hydrogen ions (in absolute units, not percentage) will be reduced to preserve the required charge balance. Second, one must assume that other ions, especially neutralizing ions such as calcium and ammonium, will remain constant. Both assumptions are reasonable simplifications given the current chemical constituents in rainfall.

The third assumption one must make is that the monitoring data are accurate and meteorologically representative. This assumption applies to the calculations for all four target values. Errors in sampling and chemical analysis lead to uncertainty in the position of the lines showing percentage reductions of wet sulfate in Figures 14 to 19. These errors, which might be

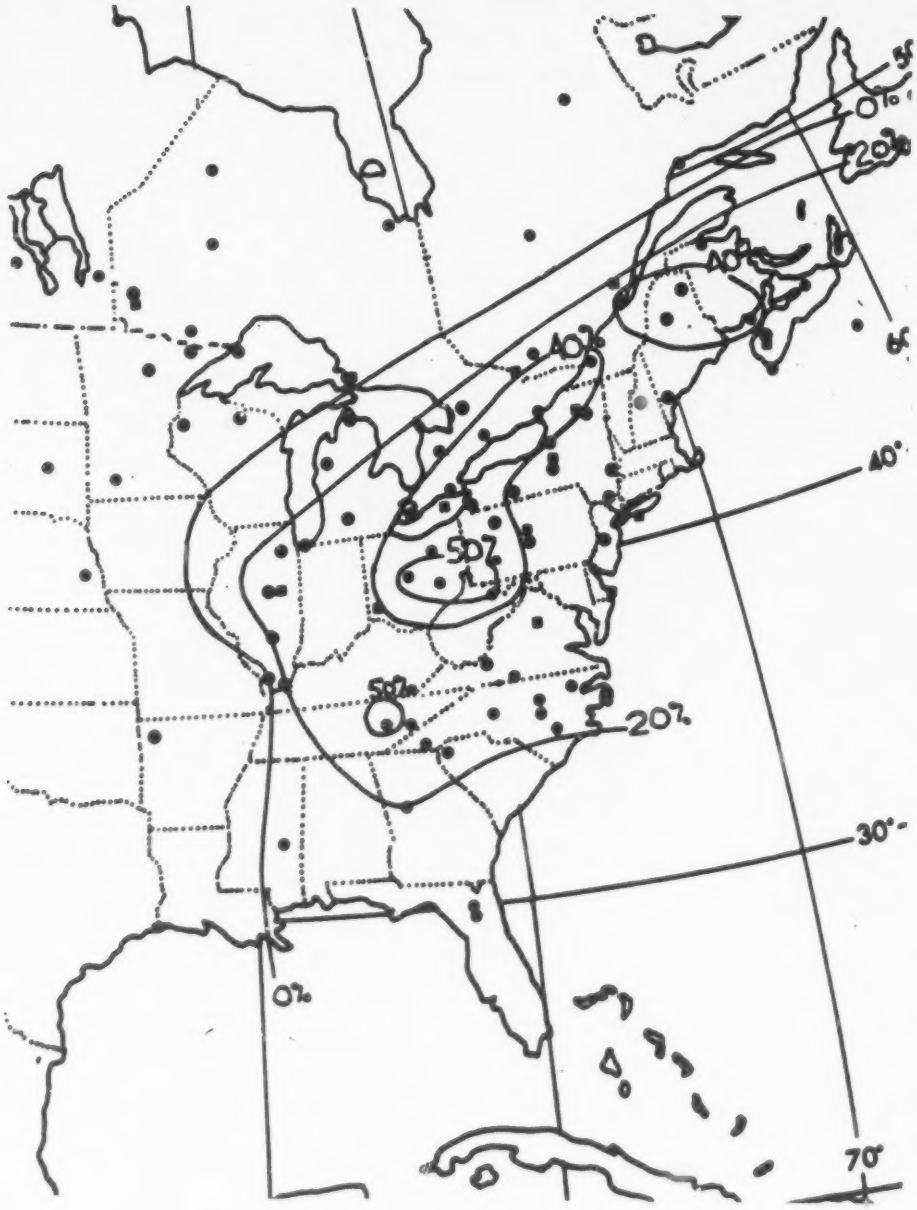


Figure 14: Percent reduction in wet sulfur deposition so that wet sulfate is reduced to 20 kg/ha or lower (1980 data).

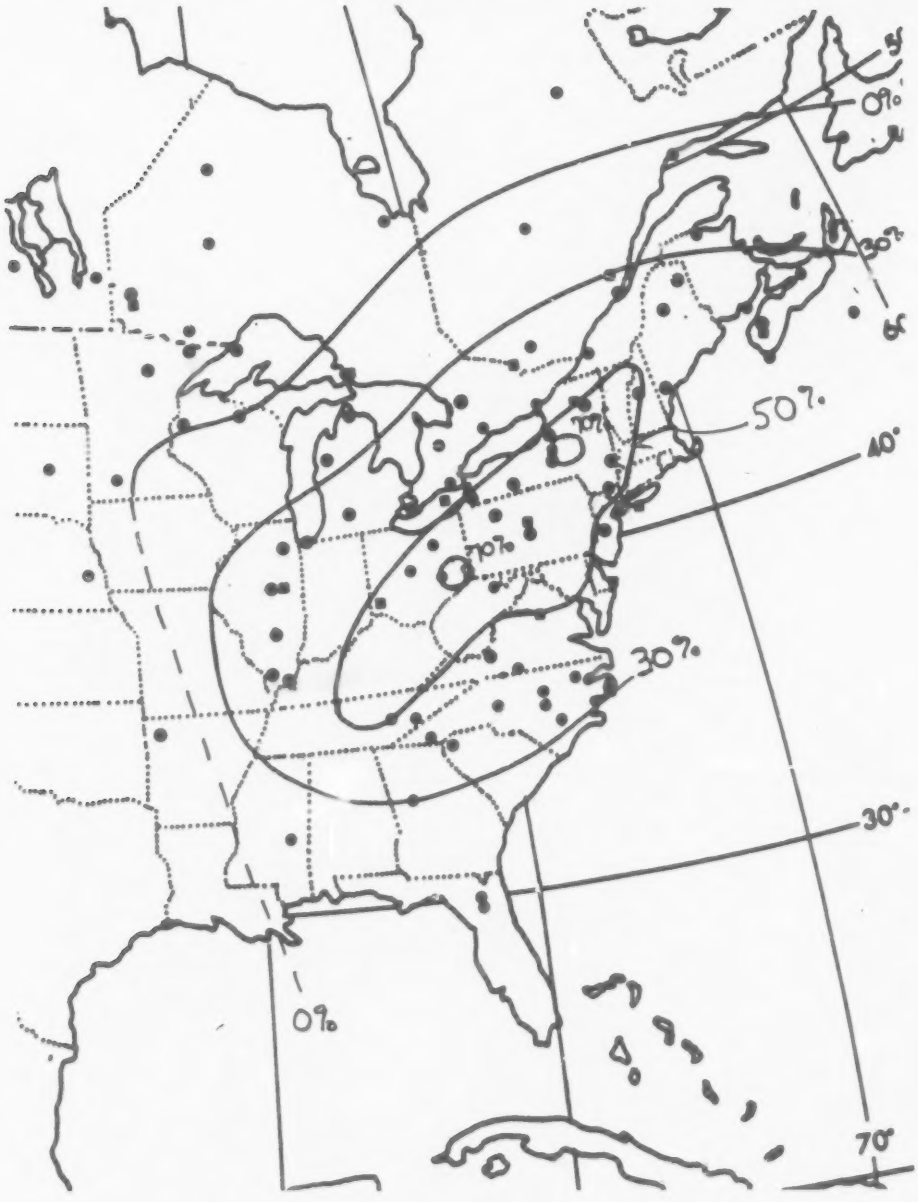


Figure 15: Percent reduction in wet sulfate deposition to increase pH to 4.6 or greater (1980 data).

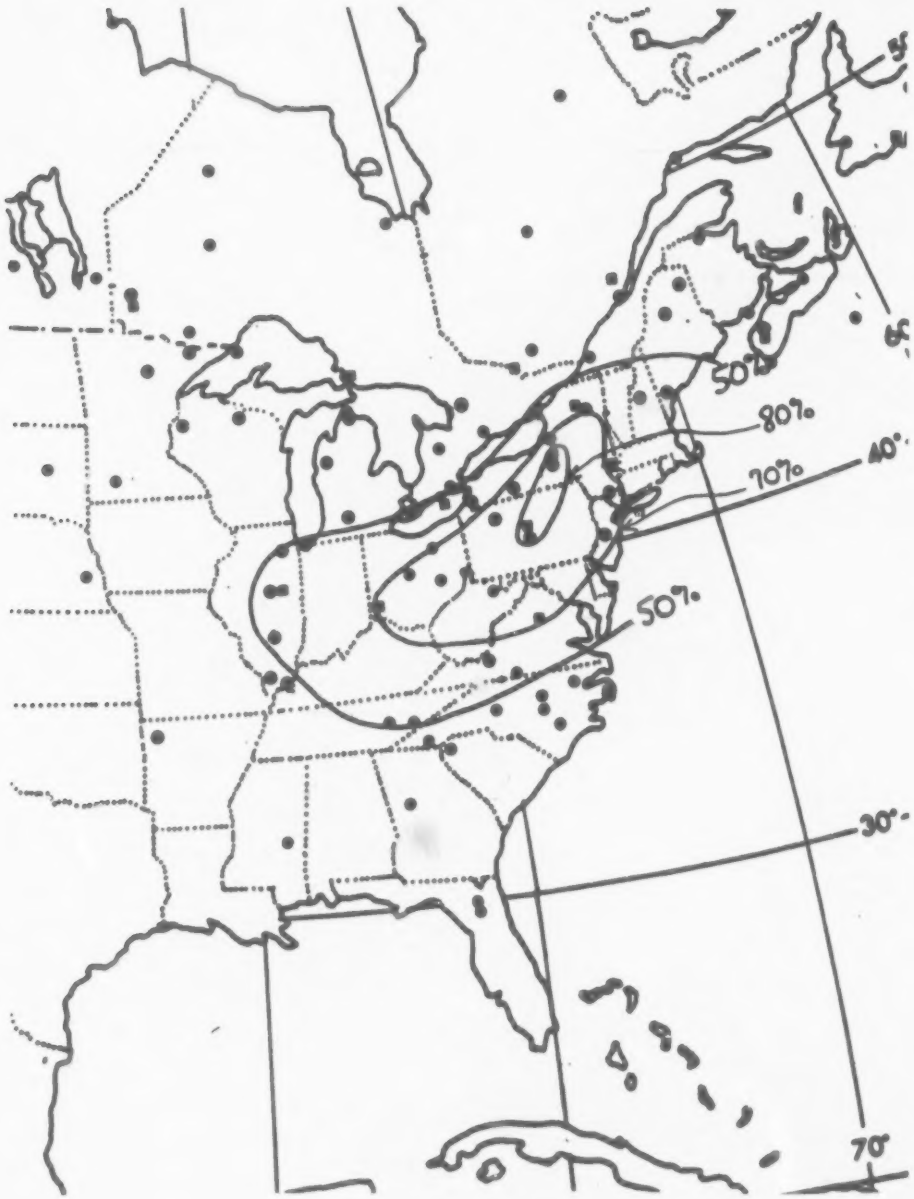


Figure 16: Percent reduction in wet sulfate deposition to increase pH to 4.7 or greater (1980 data).

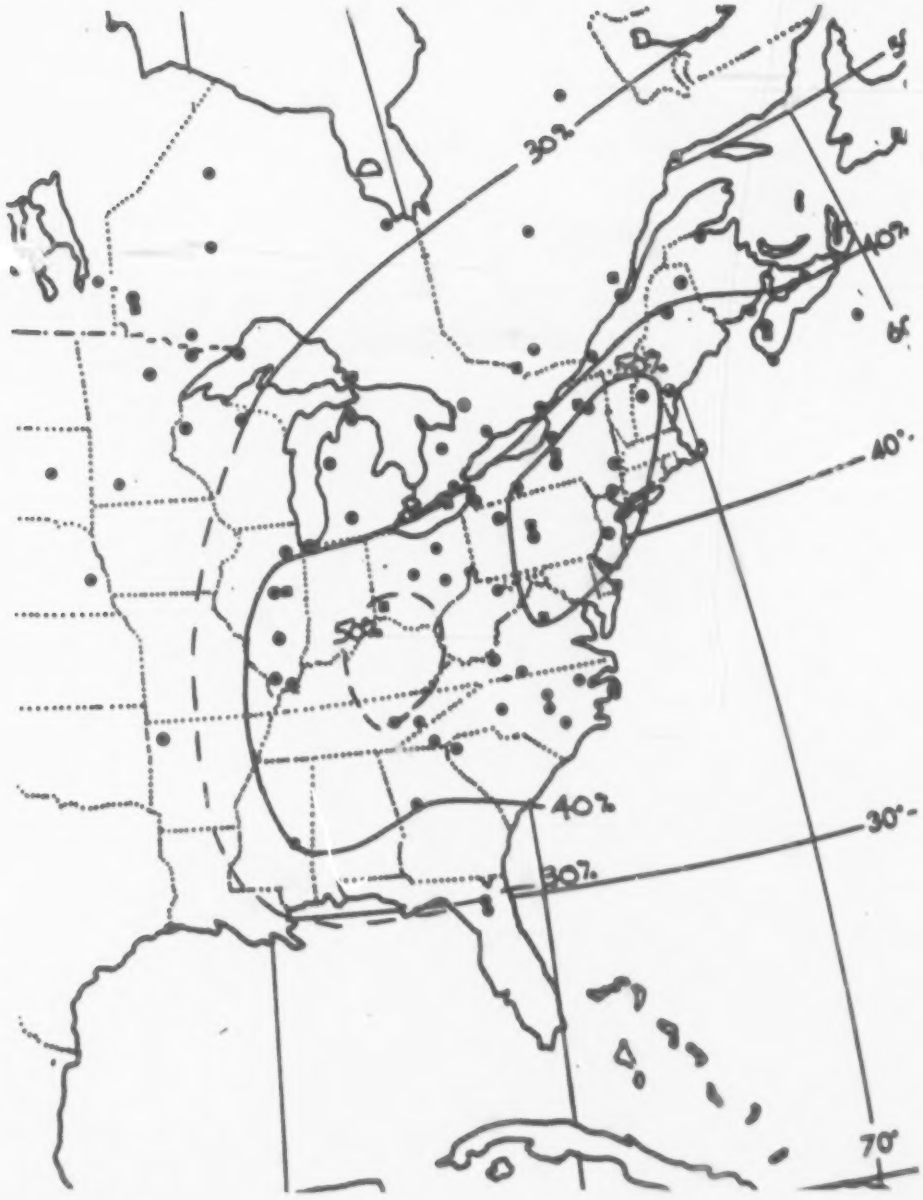


Figure 17: Percent reduction in wet sulfate deposition so that H^+ deposition is reduced by 50% (1980 data).

FIGURE 18: PERCENT REDUCTION IN SULFATE DEPOSITON TO INCREASE PH OF ACID RAIN TO 4.7 OR GREATER. (1979-80 NADP DATA)

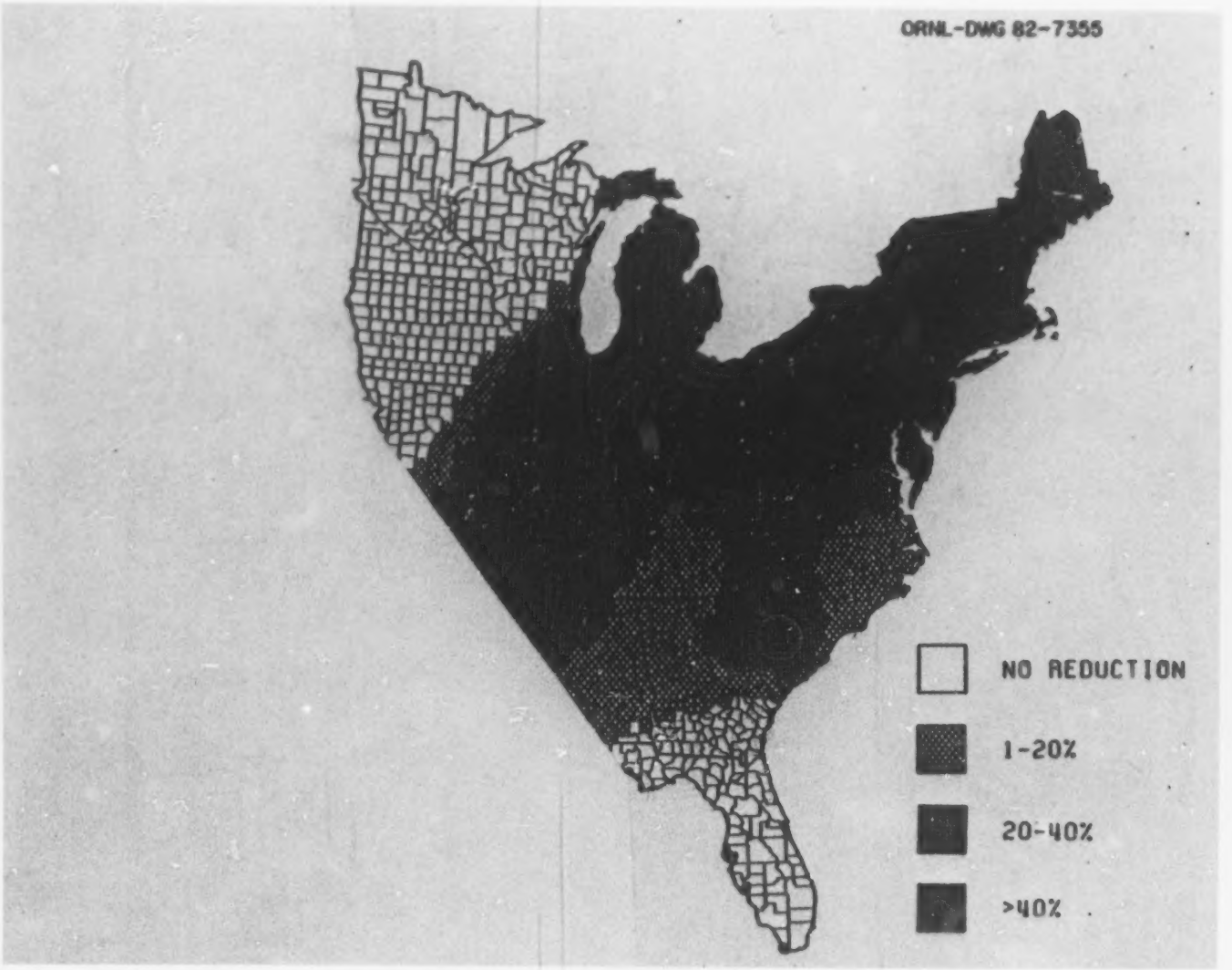
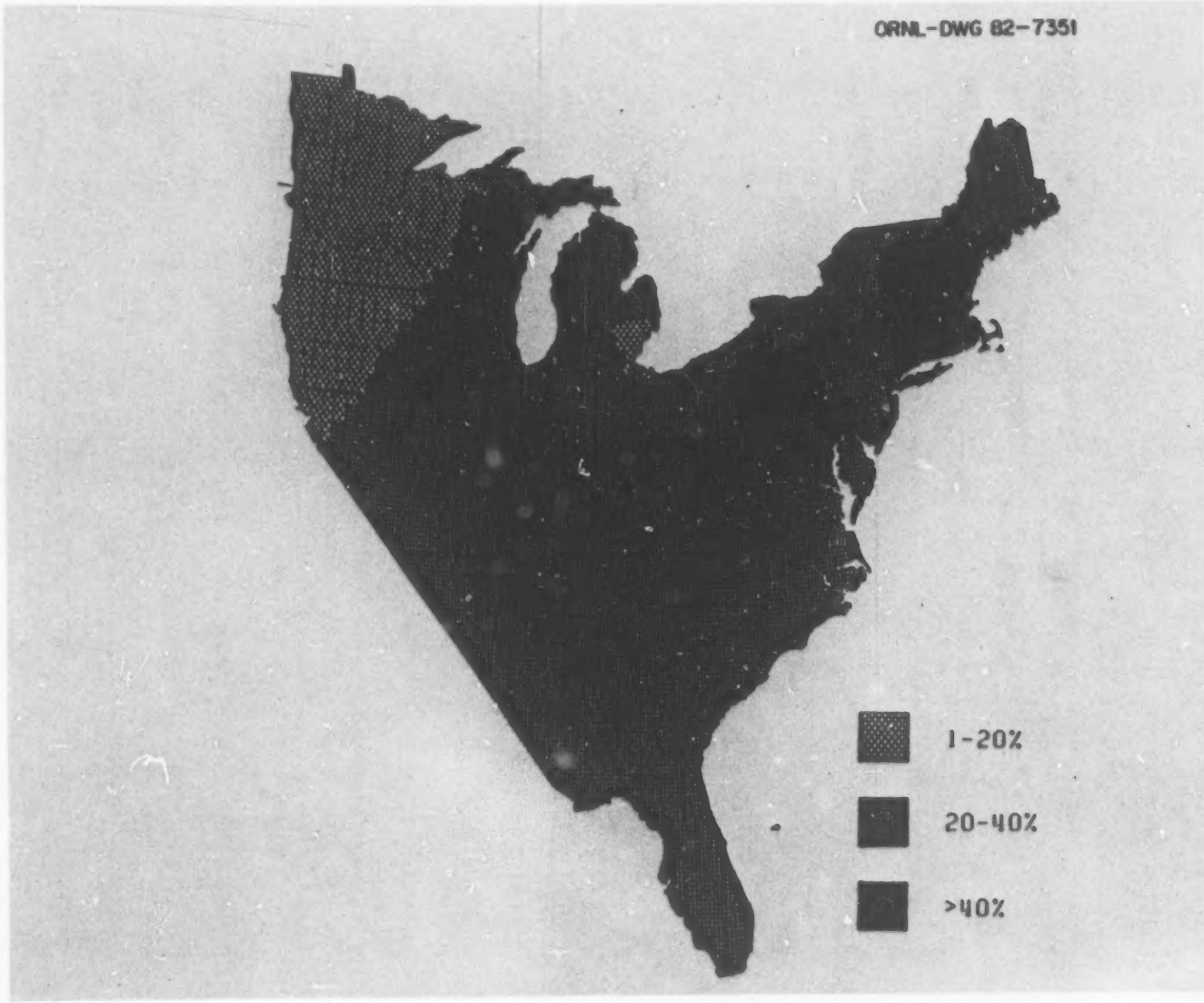


FIGURE 19: PERCENT REDUCTION IN SULFATE DEPOSITION SO THAT H+ DEPOSITION IS REDUCED BY 50% (1979 80 NADP DATA).



on the order of 10%, would translate into a positional uncertainty ranging from 50 to several hundred kilometers.

Year-to-year weather variations would also shift the position of the lines on the maps. Figures 14 to 17 are derived from 1980 data -- a year in which the annual precipitation was about average in most regions. However, data for other years would not be expected to be identical. For comparison, maps of wet sulfur reductions required to meet two of the targets for one year covering most of 1979 and part of 1980 (figures 18 and 19) are included. How much of the substantial differences are due to errors in sampling, the smaller number of sampling stations operating during the earlier period, or different weather patterns is impossible to say.

Figure 14 presents a map of regional wet sulfur deposition reductions to meet the first target value (20 kg/ha wet sulfate). Figure 14 shows that to reach this target, wet sulfur deposition would have to be reduced by about 50% in the areas of heaviest deposition, and by over 30% across broad regions of the eastern U.S.

Figures 15 and 16 show the wet sulfate deposition reduction needed to meet target values of average rainfall pH of 4.6 and 4.7. The more stringent target of pH 4.7 requires reductions of over 70% in areas of highest deposition. The less stringent target of pH 4.6 would require about a 10% smaller reduction in the peak deposition areas; the required reduction drops by a larger percentage in areas of lower deposition.

Figure 17 displays wet sulfate deposition reductions to meet target 4, a 50% reduction in hydrogen ion concentration in precipitation. The peak reductions required are on the order of 50%, with broad regions of the eastern

U.S. requiring wet sulfate reductions of 40% to reach this target.

Figures 18 and 19 display reductions required to meet target values 3 and 4, but using 1979 instead of 1980 data. For both calculations, the 1979 data show lower required reductions, with the peak reductions shifted to the northeast. Again, these differences are a combination of the variation one would expect from year to year, and the limitations of the data used in the calculations.

As figures 14 through 19 illustrate, the range of reductions required to reach each of the four targets -- and the same target in two successive years -- varies considerably. However, by comparing these figures, some qualitative statements can be made about the wet sulfate reductions needed to reach the targets suggested by the various groups listed above.

In regions of highest deposition, target deposition reductions range from 50-80%. For New England, deposition reductions required to reach the targets vary from about 30 to 70%. For sensitive areas south of the peak deposition region (such as the southern Appalachians in Tennessee), wet sulfur deposition reductions needed to reach the targets are on the order of 20 to 50%. In the upper Midwest (e.g., the lake regions of Wisconsin and Minnesota), deposition would have to be reduced by 0 to 40% to reach the targets.

In the previous section, the uncertainties inherent in model-based predictions of deposition reductions resulting from emissions reductions were characterized. Plausible (though uncertain) model-based estimates in both wet and dry sulfur deposition reductions that might be achieved by an 8 to 10 million ton per year reduction of sulfur dioxide emissions (as required by S. 1709 and S. 1706) were presented.

Because the deposition targets are expressed as reductions of wet sulfur deposition, and due to the large uncertainties in both sets of analyses, only tentative comparisons can be made between the deposition reductions that might result from reductions of sulfur dioxide emissions, and the deposition reductions required to meet the deposition targets to protect sensitive resources. In areas of highest deposition, for example, Western Pennsylvania, the suggested target reductions to protect sensitive resources might not be achievable with an 8 to 10 million ton reduction of sulfur dioxide emissions; in areas of lower deposition, such as northern New England, the southern Appalachians and the upper Midwest, the target deposition reductions might be achievable with sulfur dioxide emissions reductions of this magnitude.

The uncertainties about reaching these targets are greater in those areas that receive most of their sulfur deposition in wet form (for example, the Adirondack Mountains). As discussed in the previous section, for a given level of sulfur dioxide emissions reductions, the reduction in wet deposition may be less than for dry deposition. Moreover, changes in wet sulfur deposition from a given change in sulfur dioxide emissions may be dependent on concurrent levels of co-pollutants (such as nitrogen oxides and reactive hydrocarbons).

M. LEGAL MECHANISMS FOR THE CONTROL OF INTERSTATE AND TRANSBOUNDARY AIR POLLUTION

Summary

Long-range transport can carry air pollution across state lines and national boundaries, and transform air pollution control policies for individual states into issues of national and international concern. Much attention has been focused on Congressional proposals for revising the Clean Air Act (CAA) to control long-range transport air pollutants (LRTAPs). The existing Act provides the statutory basis to implement programs to control interstate and transboundary pollution related to LRATPs; it also provides a means of remedy for interstate pollution not adequately regulated by control programs. The Act does not explicitly provide a means of remedy for transboundary (international) pollution; however, actions taken under the previous administration may have created a right to legal recourse. In addition, avenues other than the Clean Air Act could potentially be pursued to remedy transboundary pollution.

I. INTERSTATE POLLUTION CONTROL AND THE CLEAN AIR ACT

Actions taken under the interstate provisions of the Clean Air Act suggest that the Act has been interpreted by EPA and the courts to provide adequate remedy for air pollution from local sources. However, considerable uncertainty exists over how EPA and the courts will interpret these provisions with regard to LRTAPs. EPA currently takes the position that analytical techniques addressing LRTAPs are not reliable for regulatory purposes; the courts have yet to rule on this issue. The statutory language of the interstate provisions preclude direct control of acidic deposition.

A. The NAAQS SIP Process

The Act requires that states adopt implementation plans primarily designed to attain and maintain national air ambient quality standards (NAAQS) within their borders and to prevent significant deterioration (PSD) in areas which enjoy air quality better than NAAQS. Section 110(a)(2)(E) also specifically requires that state implementation plans (SIPs) include provisions which prohibit any stationary source from emitting pollutants which would prevent attainment and maintenance of NAAQS or interfere with PSD measures in another state. The Administrator may not approve a SIP which does not comply with these interstate control requirements.

B. Utility of the Interstate Provision for the Control of LRTAPs

1. Limitations of Section 110(a)(2)(E)

Section 110(a)(2)(E) has limitations for controlling interstate pollution caused by LRTAPs. The section applies only to stationary sources; it does not control mobile-source emissions. It applies only to air pollution, and offers no direct means of controlling acidic deposition. It gives little guidance to the Administrator as to how much interstate air pollution is prohibited, and does not outline the level of proof of causation required to substantiate regulation by EPA pursuant to section 110(a)(2)(E). As written, the section leaves open such questions as 1) whether pollution must be linked to individual sources to justify control, or 2) whether analytical techniques (including models), which establish the relationship between the emissions and

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air quality of "source" and "receptor" regions respectively, can be used to justify regulation of individual sources.

2. The Interstate Provision as Applied by EPA

Since the enactment of section 110(a)(2)(E) in 1977, EPA has initiated no review of state SIPs for compliance with the provision. It has issued no regulations to spell out how a state can determine whether its SIP complies with 110(a)(2)(E). EPA has reviewed and approved a number of revisions to SIPs since 1977, but the agency has not articulated definitive policies for determining whether an individual source relaxation complies with section 110(a)(2)(E). Current agency procedures, however, limit the scope of SIP review to that portion of this plan undergoing revision. Hence, where SIP revisions propose emissions relaxations for individual plants, EPA considers only the air quality impact resulting from the change in emissions for these plants.

EPA has also adopted the position that there are no adequate tools to assess long-range transport effects, and EPA practices reflect this position. First, in reviewing emissions limitations with respect to the interstate provisions, the agency only considers impacts which can be estimated by models approved pursuant to the agency's modeling guideline. This effectively limits consideration of impacts to within 50 km of the source, because there are no approved models to estimate long-range transport impacts. Second, the agency does not consider potential impacts of the transformation products of the pollutant for which a SIP relaxation is being considered. For example, the agency reviews the interstate effect of a sulfur dioxide (SO_2) relaxation, solely for its impact on SO_2 concentrations in downwind states. However, SO_2

emissions can be transformed to particulate sulfates once in the atmosphere. Sulfates contribute to total suspended particulates (TSP) for which there are national air quality standards and can cause visibility degradation. EPA does not consider the effect of an SO_2 emission relaxation on total suspended particulate levels or on visibility.

C. Remedies for Interstate Pollution under the Clean Air Act

The Clean Air Act currently contains two means of remedy for interstate air pollution. The first relies on general judicial review provisions which allow litigants to challenge EPA actions. The second initially relies on administrative review provisions under section 126 which allow states and political subdivisions to petition EPA on matters dealing specifically with interstate pollution. Since 1977, a number of suits and petitions have been filed under these provisions of the Act to remedy interstate air pollution. The actions before the courts and EPA can roughly be categorized according to the type of pollution they seek to remedy.

Early actions sought remedy for interstate air pollution in situations where pollutants were traceable to individual sources in the near vicinity. In later actions, attention has been turned from the "local source problem" and focused on interstate pollution allegedly caused by long-range transport.

1. Judicial Remedies: Section 307 Suits Involving the LRTAP Issue

States and other litigants seeking remedies for interstate pollution caused by LRTAP have challenged, under section 307 of the Act, EPA approval of SIP revisions for single sources. In the course of these suits, litigants

have attempted to obtain judicial review of EPA's refusal to address regional emissions through the SIP process. They have also attempted to compel EPA to use models and other techniques to assess long-range transport effects.

Litigants have advanced the following reasons for challenging EPA approval of the SIP revisions: 1) the Administrator failed to properly review the LRTAP effects of the individual source for which a relaxation was approved because EPA relies only on "approved models" to estimate impacts within 50 km of the source; 2) EPA failed to consider the air quality impacts of the transformation products of SO_2 [1/]; and 3) the Administrator failed to review the whole SIP to determine whether the cumulative emissions of sources governed by the SIP does not cause pollution in violation of 110(a)(2)(E).

2. Section 126: The Administrative Petition Process

Section 126 is a companion section to 110(a)(2)(E). It requires states to provide notice to nearby states of sources which "may significantly contribute" to air pollution in excess of NAAQS. In addition, it provides an administrative remedy for interstate pollution through petition to the Administrator of EPA. The section provides in part that a

> . . . State or political subdivision may petition the Administrator for a finding that any major source emits or would emit any air pollutants in violation of the prohibition of section 110(a)(2)(E)(i).

1/ Petitioners challenge EPA's position by asserting that: 1) there are NAAQS for particulates; 2) SO_2 is transformed to sulfates which are particulates; and that therefore, 3) SO_2 emissions must be controlled to ensure they don't prevent attainment and maintenance of NAAQS or PSD increments for TSP.

Within 60 days of receipt of a petition, the Act requires the Administrator to hold a public hearing and either make such a finding or deny the petition.

a. Petitions under Section 126 Seeking Remedy for LRTAPs

States seeking remedy of interstate pollution caused by the long-range transport of emissions from multiple sources have filed section 126 petitions in tandem with suits under section 307. At least nine petitions of the State of New York, and the petitions of Pennsylvania and Maine, have been consolidated into a single proceeding, which places before EPA the gamut of issues involved in the long-range transport controversy.

The consolidated petitions claim that SO_2 and TSP emissions of sources governed by SIPs in the states of Illinois, Indiana, Kentucky, Michigan, Ohio, West Virginia and Tennessee are causing interstate pollution in violation of 110(a)(2)(E). The petitioners allege that such pollution prevents attainment and maintenance of NAAQS for both TSP and SO_2 and interferes with PSD requirements. Petitioners further allege that the conversion of SO_2 to particulate sulfates contributes to the aforementioned problems, and, in addition that sulfates present a hazard to health and cause acid rain, which results in damage to aquatic and terrestrial ecosystems. Maine specifically claims that conversion of SO_2 emissions causes visibility degradation in the mandatory Class I area of Acadia National Park in violation of the Act. Petitioners have asked that EPA review the SIPs for long-range transport effects and have requested major reductions in emissions in the eastern United States.

Petitioners have presented extensive information on: the air quality problems and alleged environmental problems resulting from acid rain in their states; air quality data and modeling results which allegedly demonstrate that interstate pollution contributes significantly to these problems; air quality models and analytical techniques available to assess the long-range transport of air pollutants; the relative stringency of emission controls between the midwestern states and the petitioning states; and finally, the alleged inequities arising between the midwestern states named in the petition and the states of New York, Pennsylvania, and Maine as a result of the different levels of emissions limitations which petitioners claim are needed to control interstate pollution.

Extensive comments have been filed concerning the information presented by the petitioners and the issues raised. As yet, EPA has issued no preliminary findings in the proceeding.

D. Review and Remedies under 307 SIP Challenges and 126 Petitions Dealing with LRTAPs

The two means of remedy for interstate pollution -- judicial and administrative -- may differ as to the type of review which can be obtained. EPA has asserted, and the Second Circuit Court of Appeals has ruled, that EPA is not required to review the whole SIP for its compliance with the Act when EPA reviews proposed SIP revisions which only amend portions of the plan. Accordingly, plaintiffs might not be able to obtain review of an implementation plan as a whole in the context of SIP revision for individual sources. Thus, the section 126 administrative proceeding may be the only way

for states to remedy pollution caused by the cumulation of emissions from all stationary sources regulated by a SIP.

It is not clear what relief the petitioners could win if successful in the suits pending before the courts or in the section 126 proceedings before the agency; however, a wide range of results are possible. Neither route is a means to compel direct control of acidic deposition. Litigants under section 307 seeking EPA review of SIP relaxations might obtain court decision requiring the agency to consider long-range transport effects using "state-of-the-art" techniques. The scope of the New York-Pennsylvania-Maine section 126 petitions is wider; the petitioners could, if successful, win a reversal of EPA SIP review policies, and EPA could require broad-scale reductions in emissions of SO_2 and TSP in midwestern states.

As appeals from any determination reached by EPA on the New York-Pennsylvania-Maine section 126 petition could come before the District of Columbiua Court of Appeals. Thus, in the absence of Congressional direction to EPA and the courts on long-range transport, the appeals court could become the arbiter of the long-range transport pollution controversy between the midwestern and northeastern states.

II. CONTROL OF TRANSBOUNDARY POLLUTION CAUSED BY LRTAP'S: THE U.S.-CANADIAN CONTEXT

The problems of interstate and transboundary (international) pollution control are strongly linked. Winds that transport pollutants over state lines can also carry them across the border between Canada and the United States. Mechanisms are available for cooperative solutions to the transboundary pollution issues. Means of control are potentially available through 1) a

bilateral accord, 2) under the Clean Air Act, and 3) under domestic common law and under international law. The efficacy of these mechanisms to deal with transboundary pollution depends substantially on the combined commitment of the U.S. and Canada to control such pollution.

A. Diplomatic/Legal Context of the U.S.-Canadian Transboundary Air Pollution Issue

The United States and Canada have a long history of cooperation concerning environmental affairs and joint commitments to principles for controlling transboundary pollution. Beginning with the Boundary Waters Treaty of 1909, the two countries have expended considerable efforts in negotiations and/or arbitration to settle transboundary pollution problems, most recently in negotiations culminating in the 1978 Great Lakes Water Quality Agreement. In recent years, both the U.S. Congress and the Executive Branch have made repeated commitments to bilateral research, consultation, and negotiations on transboundary air pollution, as well as to the multilateral Declaration of the U.N. Conference on the Human Environment (Stockholm Declaration 1972), and the 1979 Economic Commission for Europe "Convention on Long-Range Transboundary Air Pollution".

There is no agreement between the U.S. and Canada directly governing transboundary air pollution; however, cooperative activities have culminated in a Memorandum of Intent. Under the Memorandum the two countries have agreed to begin negotiations to reach a bilateral accord. The Memorandum also states the two countries' commitments to take control actions in the interim. Although the status of the Memorandum is unclear, it does not appear to be

legally binding on Canada or the United States. Its force derives from the good intentions of both countries.

The MOI also established five Work Groups to assist in preparing and conducting negotiations. The Work Groups published interim reports in 1981 and are, at this writing, working on the final reports originally due in January, 1982.

1. Transboundary Pollution and U.S.-Canadian Relations

The current Administration has taken the position in the bilateral negotiations that while long-range air pollution in general, and acid deposition in particular, may present serious problems, too many scientific aspects of long-range transport and its effects are inadequately understood to allow the U.S. to embark on a bilateral control program. As the State Department's Canadian affairs officer succinctly states the position, ". . . the scientific relationship between emissions and effects is not clear. We simply don't know where to put the money at the present time to control acid rain." Hence, while no eventual course of action has been formally barred, the Administration has signalled its intention to await more information before reaching a decision on bilateral accord.

The Administration also asserts that good faith efforts under the MOI do not require EPA to consider the international consequences of emission relaxations under the SIPs -- taking the position that, so long as local NAAQS are protected, the U.S. is vigorously enforcing existing laws as agreed under the MOI. Canadian officials assert that good faith efforts require the U.S. to at least "hold the line" on regional emissions, if not to reduce emissions.

The U.S. believes that reasonable progress toward reaching a bilateral accord is being made. Canada, however, perceives U.S. pollution control policies as undercutting the diplomatic efforts to reach a bilateral accord. Moreover, Canada is dissatisfied with the progress made in the negotiations and has recently threatened to withdraw from the talks altogether. Acidic precipitation resulting from the long-range transport of air pollution has emerged as one of the most significant bilateral issues between Canada and the United States.

B. Transboundary Pollution Control Under the Clean Air Act

1. Section 115

Congress adopted section 115 of the Clean Air Act to provide a means to deal with transboundary air pollution. The section is invoked by the Administrator of EPA on his own initiative or at the request of the Secretary of State. The Secretary must allege, or the Administrator must determine, that pollution emitted in the United States causes or contributes to air pollution which "may reasonably be anticipated to endanger public health or welfare in a foreign country". The section provides that the Administrator can make such a determination "upon receipt of reports, surveys or studies from any duly constituted international agency"; however, it contains no language as to how the Secretary of State will be apprised of a problem.

Once the Secretary of State has requested the Administrator to activate the 115 process or the Administrator has made a finding of "endangerment", the statute requires the Administrator to give formal notice to the governor(s) of the state(s) in which the emissions causing the pollution originate. The

Administrator must take this step, however, only if he "determines the foreign country receiving the pollution has given the United States essentially the same rights with respect to the prevention or control of air pollution occurring in that country as is given that country by [section 115]." Thus the Administrator must make a determination of reciprocity to trigger the mandatory duty under section 115.

Section 115 relies on the SIP revision process as the means to abate international pollution prohibited by the section. Section 115 provides that notice given to the governor constitute a finding with respect to section 110(a)(2)(E) -- i.e., that the state implementation plan is inadequate to comply with the requirements of the Clean Air Act and must be revised.

2. Utility of Section 115 for the Control of LRTAP

Section 115 can be construed to permit the control of transboundary pollution caused by acidic deposition. However, the language of section 115 falls short of giving the Administrator clear guidelines on how to implement the section. It does not detail the procedures for the Administrator to follow in identifying which state or states are the source of the transboundary emissions, and provides no guidance as to how to allocate control responsibilities when emissions from more than one state create transboundary problems. The legislative history of section 115 is also silent on these issues.

EPA's current practices for reviewing the state SIP control of interstate pollution would, if applied to transboundary pollution, limit the potential for controlling possible acidic deposition and air pollution in Canada. As yet, the present Administrator of EPA has not chosen to activate the

transboundary pollution control provision of the Act, nor has EPA coordinated domestic pollution control efforts to limit potential transboundary pollution.

3. Activity Under Section 115

a. The Mitchell Request

The Canadian Clean Air Act was amended by unanimous vote in both Houses of Parliament in December 1980 to provide clear authority for the Canadian federal government to take steps to control the emission of pollutants affecting another country. The amendment was designed primarily to allow "mutual recourse between Canada and the United States".

On December 23, 1980, Senator George Mitchell of Maine sent letters to the Administrator of EPA and the Secretary of State calling their attention to the recently enacted legislation, and to reports prepared by the International Joint Commission (IJC) and the U.S.-Canada Research Consultation Group on Long-Range Transboundary Air Pollution (RCG). The Senator reviewed the information in the reports and concluded his letters with a strongly-worded request that action be taken under section 115.

b. The Costle Response: Determination to Activate Section 115

On January 13, 1981, Administrator Costle of EPA responded to the Mitchell request and sent letters to both the Senator and the Secretary of State announcing his findings. He found, based on his review of the IJC reports and the Clean Air Act, that section 115 could be activated to control acidic deposition in Canada. He also determined that the Canadian Clean Air

Act provides Canada with authority to give the U.S. essentially the same rights as Canada under section 115, and that at present Canada was so interpreting the Act. He noted that this second aspect of EPA's determination is necessarily a dynamic one and would continue to be influenced by Canadian actions. Thus, having made the requisite determination to activate section 115, the Administrator instructed EPA staff to begin work to identify which states should receive formal notification, and to lay the groundwork to assist these states in appropriately revising their SIPs.

There have been no indications that the work initiated by Administrator Costle is proceeding under the present EPA Administrator. As yet, section 115 notification has not been given to any state, and EPA appears not to have continued the process leading up to notification. EPA has issued no statement, however, which reverses the Costle determination. No court has yet reviewed the legal significance of Costle's action.

C. Remedies for Transboundary Pollution

1. Recourse under the Clean Air Act

The initial decision to activate section 115 is clearly discretionary. Thus, the statute does not explicitly provide a cause of action which Canada or others could pursue in domestic courts to remedy transboundary pollution. An argument can be made, however, that the actions of former Administrator Costle "activated" section 115 and created a legal obligation for the present Administrator to revise SIPs to control acidic deposition in Canada.

Section 304 of the Clean Air Act authorizes "any person" to bring suit to compel the Administrator to perform non-discretionary duties under the Act.

It is clear that under the citizen suit provisions a number of different groups could bring suit to compel action under section 115 -- among them are environmental organizations and northeastern states that might "benefit" from activation of the section. The eligibility of Canada or the Province of Ontario to bring suit is less clear. Section 302(e) does not specifically mention foreign governments and it is an unsettled question whether foreign governments are considered to be "persons" under section 302(e) and 304. However, standing to sue would not be a barrier to litigation by plaintiffs having interests similar to Ontario or Canada.

Should a suit be initiated under section 304 to compel the present Administrator to notify states under section 115 to revise their SIPs, a court could be faced with the question whether Costle's determinations were arbitrary and capricious, or in excess of statutory authority and, as such, unlawful. The present Administrator would have the burden of convincing a court that the previous Administrator's determinations were invalid. Since the statute gives the Administrator great discretion, and courts as a rule pay great deference to administrative determinations, this would be a difficult burden to overcome.

2. Resource Through Diplomatic Channels

The Canadian Government could choose to link the transboundary issue to other areas of bilateral concern in an effort to encourage adoption of policies to reduce the long-range transport of air pollution. Canada is an important ally and neighbor. The U.S.'s involvement with Canada is probably greater than with any foreign country. The two-way trade is about $77 billion, which is greater than that between the U.S. and all the countries of

the European Community. The two countries are allies in NATO and have a unique military joint command, NORAD, the North American Air Defense Command.

3. Recourse Under Domestic Common Law and International Law

In theory, domestic common law and international law also provide a means of legal recourse should Canada or others be dissatisfied with U.S. control efforts. However, in practice domestic common law may not, and international law does not, provide an effective means to compel U.S. control efforts. In the case of domestic common law, recent court decisions as well as problems surrounding proof of causation may limit the utility of this avenue; in the case of international law, there are no effective means to enforce legal doctrines.